Effects of urbanization on the biological integrity of Puget Sound lowland streams:
Restoration with a biological focus

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Abstract

Effects of urbanization on the biological integrity of Puget Sound lowland streams:
Restoration with a biological focus

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Rapid urbanization threatens the condition of streams and rivers across the Pacific Northwest. Efforts to protect and control urban streams have traditionally focused on physical channel conditions and chemical water quality. This study is designed to use biological attributes of these streams—measured with the multimetric index of biological integrity (B-IBI) based on benthic macroinvertebrates—to assess stream condition. Between 1997 and 1999, invertebrate samples were collected from 45 sites in second and third order streams of the Puget Sound lowlands of Western Washington. The locations of 14 sites were chosen to evaluate the placement of large woody debris (LWD) as a restoration technique on five streams. Urbanization was characterized by a 1998 satellite land cover classification and was measured across several spatial scales. The relationships among metrics of the B-IBI and stream substrate and hydrologic features were also evaluated at a sub-set of sampling sites. B-IBI declined as urban land cover increased both across the entire basin and within riparian zones. The effectiveness of localized patches of riparian corridor in maintaining biological integrity varied as a function of the percentage of urban land cover in the sub-basin. Channel roughness and hydrologic flashiness were also correlated with B-IBI. Below the five restoration projects, there was no overall detectable improvement in B-IBI. The aquatic biota is sensitive to a variety of urban impacts controlled over both large and small spatial scales. Restoration efforts that deal with only one local impact type without addressing larger scale issues are unlikely to successfully restore the biota of degraded streams.

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INTRODUCTION

Urban sprawl threatens the biological health of Pacific Northwest streams and rivers. A similar story is unfolding across the United States; metropolitan areas now cover over 19% of the U.S. and house over 75% of the population (Stoel 1999). Between 1970 and 1990, rural lands were converted to residential and commercial centers at the rate of 400,000 acres per year (American Rivers 1999). As our national landscape shifts increasingly towards pavement and manicured lawns, streams disappear completely underground into culverts. Those urban streams that still run above ground are typically highly engineered channels designed more for flood control and sediment transport than ecological considerations (Roesner 1997). Within the field of stream ecology itself, very little study has focused on urban stream systems. Not surprisingly, the way in which we've historically managed urban streams and rivers has also tended to ignore biology (Karr and Chu 1999b, Yoder and Rankin 1998).

Evidence of the biological degradation of streams across North America is overwhelming. Thirty-six percent of river miles surveyed for the 1994 National Water Quality Inventory failed to support healthy aquatic communities (U.S. EPA 1995), and recent studies find temperate freshwater species extinction rates as high as for tropical forests (Ricciardi and Rasmussen 1999). In North America, 67% of unionid mussels, 51% of crayfish, 40% of amphibians, and 37% of fish species are extinct or imperiled (Master et al. 1998). In the Pacific Northwest, salmon that historically defined this region culturally, ecologically, and economically are in sharp decline (Nehlsen et al. 1991, National Research Council 1996). On March 16, 1999 the National Marine Fisheries Service added nine populations of Pacific salmon and trout to the endangered species list—the first time such protection has extended to major metropolitan areas of the Pacific Northwest (Gorman and Sears 1999).

One response to these listings has been increased focus on urban stream restoration[1].

[1] The term "restoration" is used liberally in this study to refer to all channel alteration, habitat enhancement, and rehabilitation efforts aimed at improving overall stream condition. Restoration in the literal sense of the

Over the last ten years more than 300 restoration projects were installed in urban areas around the Puget Sound lowlands of Washington State alone (Kropp 1998). With millions of dollars in federal funds recently allocated for salmon recovery (Mapes 1999) and a public increasingly active in river conservation and restoration (Karr et al. 2000, Riley 1998), this number is likely to rise rapidly. That is a good thing. What is worrisome is a deficiency of consistent pre- and post-project monitoring to guide project placement and design, and to evaluate what techniques are working where. The mission underlying the majority of these projects is salmon recovery, yet very rarely are salmon or any other element of stream biota directly monitored to assess restoration success. Of the 300 plus projects cited above, less than five percent were evaluated with any sort of biological data (Kropp 1998). In order to truly "restore and maintain the chemical, physical, and biological integrity of this Nation's waters (Clean Water Act 1972)," we need to pay more attention to the biota that inhabit those waters. The tools and techniques for doing so are now well established (Karr and Chu 1999a), and the benefits well documented (Davis and Simon 1995, Simon 1999). Biological assessment should play a more central role in water resource management.

Study framework and objectives

The overall objective of this study is to advance the use of biological monitoring in urban stream management and restoration. The specific method of biological assessment applied is the benthic index of biological integrity (B-IBI), a multimetric index based on attributes of the benthic invertebrate community (Fore et al. 1996, Kerans and Karr 1994, Kleindl 1995). Although utilized by water resource agencies for routine stream monitoring, B-IBI has yet to be widely applied towards targeting and/or evaluating restoration efforts. In order to use this index most effectively for such assessment

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word ("…bringing back into a former, normal, or unimpaired state or condition." Webster's New World College Dictionary, third edition, 1997) is often infeasible in urban basins due to the extent of irreversible land cover modification.

purposes, it is essential to understand how B-IBI responds to modification of both land cover and channel form/function. This study therefore has three components: 1) analysis of B-IBI variability relative to changing land cover 2) evaluation of the diagnostic properties of B-IBI (e.g., how do metrics of this index respond to different channel impact types?), and 3) assessment of biological response associated with in-stream restoration projects. For the land cover analysis, a recent satellite classification (Botsford et al. 1998) is used to examine B-IBI response across gradients of urbanization. This GIS-based analysis is conducted over three spatial scales: the entire drainage basin, the riparian corridor, and the local area upstream of invertebrate collection sites. For the diagnostic component of this study, relationships between B-IBI and metrics to substrate and hydrologic stream features are measured. Assessment of restoration projects focuses on the placement of large woody debris (LWD), a common restoration technique in Pacific Northwest streams (Booth et al. 1997, Larson 1999). Three broad questions emerge as the focus of these analyses:

1. How do extent and scale of urbanization across stream basins influence B-IBI?
2. How do metrics of B-IBI respond to modification of stream flow and substrate?
3. Does placement of LWD in urban streams improve downstream biological condition?

Background on biological integrity

The legal context of clean water. The Clean Water Act passed by Congress in 1972 has as its main objective, "to restore and maintain the chemical, physical, and biological integrity of this Nation's waters" {PL 92-500, Clean Water Act (CWA), §101(a)}. Efforts to protect and control urban streams have focused predominately on chemical water quality and physical channel conditions (i.e., "habitat") as surrogates for biological integrity (Karr 1991, Yoder and Rankin 1998). Because declining biological conditions in running waters have many potential causes (e.g., water quality, habitat structure, flow regime, energy source, and biotic interactions), a broader perspective is needed (Karr and Chu 1999a: Table 9). Monitoring approaches that focus only on the sources of biological degradation (the stressors) rather than directly measuring the biota (the response), often substantially underestimate degradation to streams and rivers (Davis et al. 1996, Yoder and Rankin 1998). Recognizing this, more than 31 states have adopted narrative or numeric biological criteria into their water quality standards over the past decade (Davis et al. 1996). The U.S. Environmental Protection Agency (EPA) has made it a priority that biological criteria be a key component in the water quality programs of all 50 states by the year 2005 (U.S. EPA 1998).

The history of biological monitoring. Over the last century, biological monitoring tools needed to establish water quality criteria have taken a number of different approaches and examined a wide range of taxa—most commonly, fish, invertebrates, and algae. Biological assessment (measuring and evaluating biota directly) has ranged from saprobien indexes (Kolkowitz and Marsson 1908, Hilsenhoff 1982), to toxicity testing (Buikema and Voshell 1993), indicator species abundance (Farwell et al. 1996), diversity indexes (Wilhm and Dorris 1966), and more recently to multivariate models (Parsons and Norris 1996, Wright et al. 1993) and multimetric indexes (Karr 1981, Karr and Chu 1999a). For a review of the history of biological assessment in U.S. waters see Davis (1995). One of the most common approaches today is the multimetric index, currently in use in over 42 U.S. states and numerous countries (Karr and Chu 1999a). The multimetric approach evaluates biological condition by integrating measures of an empirically tested set of biological attributes. This approach was first used with fish in small warmwater streams of the midwestern U.S. with the index of biological integrity (IBI; Karr 1981), and has since been modified for a variety of regions and taxa. One of the many advantages of the multimetric approach is its relative simplicity; results are easily communicated and understood by non-scientists (Steedman 1988). This is a very important consideration in a field such as urban stream restoration, which is shaped as much by current policy and public opinion as by scientific study.

The benthic index of biological integrity (B-IBI). Since the mid 1990's, university scientists, water resource managers, and volunteers have used the multimetric B-IBI to evaluate the biological condition of Pacific Northwest streams with benthic macroinvertebrates (Fore et al. 1996, in press; Karr and Chu 1999a; King County 1996; Kleindl 1995). Benthic macroinvertebrates are particularly well suited for biomonitoring: they are diverse and abundant, sensitive to human disturbance, and are excellent indicators of stream condition in that they are key components of the aquatic foodweb, often long-lived, and not migratory or artificially stocked (Fore et al. 1996, Rosenberg and Resh 1993, Vannote et al. 1980). The B-IBI is composed of ten metrics of taxa richness and diversity, population attributes, disturbance tolerance, and feeding and other habits. For a given invertebrate attribute to be included as a metric in the B-IBI, it must respond predictably along a gradient of anthropogenic disturbance (Fore et al. 1996, Kerans and Karr 1994). This dose-response relationship was tested during initial B-IBI development in the Puget Sound region (Table 1; Kleindl 1995) and has been replicated in subsequent years of study (Dewberry et al. 1999, Fore et al. 1996, Karr and Chu 1999a, Patterson 1996, Rossano 1995). When values from the ten metrics are combined, B-IBI has a range of 10 to 50 and can detect five categories of resource condition (Table 2; Doberstein et al. *In press*). Questions regarding appropriate sampling equipment and procedures (Barbour et al. 1996, Cuffney et al. 1993), sample location and seasonality of sampling (Kerans et al. 1992, Kerans and Karr 1994), and laboratory procedures and taxonomic level of effort (Doberstein et al. *in press*, Klemm et al. 1990) have been addressed in past studies (see Karr and Chu 1999: Premise 19 for review). It is not my intent to re-evaluate these questions, but to move forward and apply established B-IBI methodology to the study of urban stream restoration.

Elaboration of study questions

Land cover modification and stream condition. Human modification of the landscape from agriculture, mining, logging, and urbanization is a principal threat to

Table 1. The ten metrics of the B-IBI and their predicted and observed response to urbanization. Urbanization is measured as the percentage of total impervious area in the sub-basin (Source: May 1996). R^2 values are reported for two years of independent analysis: 1994 (n=31) and 1997 (n=16). Not all metrics are expected to respond monotonically (Karr and Chu 1999a: Premise 15).

Metric	Description	Response	1994	1997
Taxa richness & composition			R^2	R^2
total taxa	richness[1]	decrease	0.65 *	0.46 *
mayfly taxa	richness[1]	decrease	0.68 *	0.29 *
stonefly taxa	richness[1]	decrease	0.54 *	0.53 *
caddisfly taxa	richness[1]	decrease	0.40 *	0.66 *
Population attributes				
dominance[3]	relative abundance[1]	increase	0.28 *	0.12
"long-lived" taxa	richness[2]	decrease	0.03	0.42 *
Tolerance & intolerance				
intolerant taxa	richness[2]	decrease	0.25 *	0.13
tolerant taxa	relative abundance[1]	increase	0.37 *	0.48 *
Feeding & other habits				
"clinger" taxa	richness[1]	decrease	0.71 *	0.58 *
predators	relative abundance[1]	decrease	0.24 *	0.31 *

[1] mean of three replicates, [2] cumulative of three replicates, [3] of three most abundant taxa, * $p < 0.05$

Table 2. Five classes of biological condition. Modified from Karr et al. (1986).

Biological Condition	B-IBI range	General Description
excellent	46 - 50	Comparable to least disturbed reference condition; overall high taxa diversity, particularly of mayflies, stoneflies, caddisflies, long-lived, clinger, and intolerant taxa. Relative abundance of predators high.
good	38 - 44	Slightly divergent from least disturbed condition; absence of some long-lived and intolerant taxa; slight decline in richness of mayflies, stoneflies, and caddisflies; proportion of tolerant taxa increases.
fair	28 - 36	Total taxa richness reduced - particularly intolerant, long-lived, stonefly, and clinger taxa. Relative abundance of predators declines; proportion of tolerant taxa continues to increase.
poor	18 - 26	Overall taxa diversity depressed; proportion of predators greatly reduced as is long-lived taxa richness; few stoneflies or intolerant taxa present; dominance by three most abundant taxa often very high.
very poor	10 - 16	Overall taxa diversity very low and dominated by a few highly tolerant taxa; mayfly, stonefly, caddisfly, clinger, long-lived and intolerant taxa largely absent. Relative abundance of predators very low.

stream health in North America and across the globe (Allan and Flecker 1993, Karr and Schlosser 1978). Streams are hierarchical systems—highly connected both longitudinally, laterally, and vertically to the landscapes they drain (Naiman and DeCamps 1990,Vannote et al. 1980, Ward 1989). The conversion of native vegetation influences streams via a variety of processes that are controlled over many spatial scales. For example, alteration to flow regime—which affects biota both via the sheer magnitude of peak flows and by a range of indirect effects on physical channel condition (Poff et al. 1997, Richter et al. 1996)—is generally a function of basin-wide processes (Allan et al. 1997). Changes in allochthonous food sources and stream shading are associated with land cover modification at the riparian scale (Gregory et al. 1991). Local scale effects include bank erosion, often influenced by streamside soil and plant cover (Dunaway et al. 1994) and/or local land use practices (e.g., grazing; Kauffman et al. 1983). With the increasingly widespread use of Geographical Information Systems (GIS), growing numbers of watershed studies are incorporating the element of spatial scale to examine the influence of land cover/use on stream condition (Allan et al.1997, Hunsaker and Levine 1995, Roth et al. 1996, Richards and Host 1994, Richards et al. 1996, Steedman 1988, Wang et al. 1997). The results of this research have broad application to water resource management; for example, they challenge the assumption that protecting riparian areas is adequate to insure high quality streams.

I take a similar hierarchical approach in the study of Pacific Northwest streams, but use B-IBI to measure stream condition and focus specifically on the conversion of forest to urban land cover. In recent studies of urban Pacific Northwest basins, land cover modification has been expressed as the percentage of total impervious area (TIA) across a given basin (Dewberry et al. 1999, Horner et al. 1997, Kleindl 1995). Overall, B-IBI is inversely related to TIA, but with a high degree of variability at lower levels of imperviousness (Figure 1). Why is it that some sites score lower than predicted by TIA (e.g., Little Bear Creek) and others higher (e.g., Swamp Creek)? I explore two explanations: the definition of what is considered "urban" and the influence of spatial scale. Total impervious area is very useful in modeling hydrologic modification (Schueler 1994), but streams are affected by many other stressors. I test an alternative measure of land cover relative to B-IBI via a recent land cover classification of the study region intended specifically for urban watershed analysis (Botsford 1998). GIS software was used to examine to what extent urban land cover change over three spatial scales influenced variation in B-IBI between and among basins.

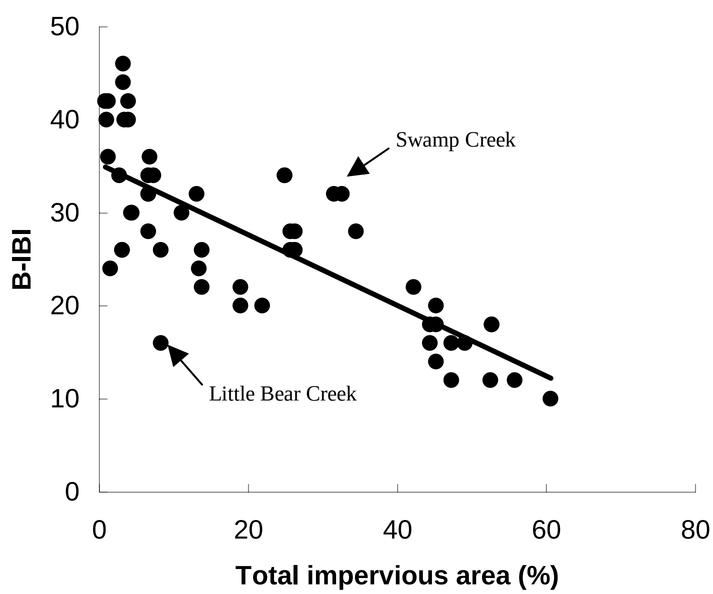

Figure 1. The relationship between B-IBI and total impervious area. This measure of basin urbanization applies to the entire sub-basin upstream of each sample point (May 1996). The dataset plotted here is for 1994 and 1995 (Karr and Chu 1999a, Kleindl 1995).

Evaluation of the diagnostic properties of B-IBI. B-IBI provides a numeric synthesis of site condition, but it can also be broken back down to derive descriptive and potentially diagnostic information from each of the ten metrics. Invertebrates vary widely in their life history requirements and tolerance to specific types of human disturbance

(Merritt and Cummins 1996, Rosenberg and Resh 1993). For instance, stoneflies generally require cool, well-oxygenated waters while many invertebrates that are classified as "clingers" prefer stable and sediment-free substrate (Merritt and Cummins 1996). Why not apply this information to diagnose specific sources of stream degradation? Two recent examples of this type of analysis come from Ohio and Japan with the concept of biological response signatures: "…biological community characteristics that aid in distinguishing one impact type over another" (Yoder 1991). In Ohio, the EPA found a strong relationship between the proportion of fish deformities (% DELT anomalies) and the relative abundance of a tolerant genus of midge (% *Cricotopus spp.*) to complex toxic impact types (Yoder and Rankin 1995). In Japan, sites receiving high volumes of domestic and agricultural effluent were dominated by tolerant midges and oligochaetes, whereas sites with high concentrations of FPOM from upstream weirs and dams had a high relative abundance of *Cheumatopsyche brevilineata*, a species of filter-feeding caddisfly (Rossano 1995).

In this study I focus on evaluating the relationships between B-IBI and metrics to flow and substrate alteration in urban Pacific Northwest streams. In the pre-development forested state, the abundant but low-intensity rainfall characteristic of this region was conveyed to streams almost entirely as sub-surface flow. In urban basins now covered largely by impervious surfaces, a shift from sub-surface to overland flow has profoundly altered the delivery of water and sediment to the stream channel (Booth and Jackson 1997, Dunne and Leopold 1978). The flow regime of developed basins commonly increases in magnitude, duration, and frequency of peak flow and decreases in summer baseflows (Booth and Jackson 1997, Hollis 1975, Richter et al. 1996). Increased overland flow also provides greater opportunity for sediment delivery to the channel, especially when there is construction activity in the basin (Lenat et al. 1981, Leopold 1968). The distribution of benthic invertebrates is strongly shaped by both adaptation to flow and substrate preference (Poff and Ward 1989, Statzner and Higler 1986, Allan 1995). What then do metrics of B-IBI indicate about flow and substrate modification in urban basins of the Pacific Northwest?

Urban stream restoration. Although urban stream restoration is increasing across the nation (Riley 1998), comparatively few examples of monitoring programs designed to measure the success of such efforts are documented (Kershner 1997). How should "success" even be defined for a diverse variety of restoration techniques applied over a variety of spatial scales? In the Puget Sound lowlands, restoration approaches range from basin stormwater detention to riparian re-vegetation to local bank stabilization (Kropp 1998). To some extent, monitoring programs should be project specific (i.e., stormwater detention efforts evaluated with hydrologic records, bank stabilization with physical channel surveys). But if the underlying goal of restoration efforts is biological, then a broader and integrative ecological measure of success is crucial (Angermeier 1997). Too often, projects are very narrowly focused on game fish (Allan and Flecker 1993), and less on overall stream condition. For instance, salmon in the Pacific Northwest are but one very important component of healthy streams. Urban streams in the region that do not currently support these fish may still support a diversity of other aquatic and terrestrial species and also serve as valuable green spaces to local neighborhoods. By losing sight of the whole picture, we risk endangering salmon further in the Pacific Northwest.

This study applies B-IBI to evaluate a specific type of in-stream restoration technique: placement of large woody debris (LWD). Much has been written about the importance of wood in Pacific Northwest streams (Bilby and Ward 1989, Naiman et al. 1992). Many urban streams of the Pacific Northwest today lack wood, as a consequence of riparian corridor development and wood removal for fish passage and flood control (Booth et al. 1997, Horner et al. 1997). Among recent urban Puget Sound restoration projects, LWD installation is second only to streamside re-vegetation (Kropp 1998). Although the placement of wood has been used with some degree of success in forested and agriculture basins (Angermeier and Karr 1984, Cedarholm et al. 1997, Hilderbrand et al. 1997), structural failure rates are often very high (Frissell and Nawa 1992). Even less is known about effectiveness in urban basins, where profound hydrologic alteration and channel confinement are potentially confounding factors (Booth et al. 1997). If LWD does stay in place, past evidence indicates that wood may serve as cover for fish and substrate for invertebrate colonizers (see above, and Benke et al. 1985). But whether or not wood placed in urban basins provides other functions as in natural streams (e.g., increasing organic matter retention, trapping sediment in pools, or attenuating high flows and thus creating a diversity of flow and channel conditions for stream biota), has not been clearly demonstrated. By monitoring the benthic communities immediately above and below LWD placement, this study attempts to detect such an overall change in stream condition.

METHODS

Study region

This study was conducted in the Puget Sound Lowland ecoregion of western Washington (Figure 2). The lowlands formed 15,000 years ago during the last glacial period between the Olympic mountain range to the west and the Cascades to the east (Waitt and Thorson 1983). Elevation ranges from sea level to 800 meters, and surficial soils are primarily glacial till and outwash (Omernik and Gallant 1986). Prior to European settlement, Puget Sound lowland watersheds were covered in dense old-growth conifer forests of western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and douglas fir (*Pseudotsuga menziesii*). Extensive riparian forests heavily shaded small streams and contributed abundant organic material to the aquatic foodchain. Large woody debris was also plentiful, and prominent in shaping the channel morphology of small streams in the Puget lowlands (Naiman et al. 1992). These streams today still support a diversity of terrestrial and aquatic life, and serve as spawning and rearing grounds for seven species of native salmonids (Kruckeberg 1991).

In the last three decades, urban sprawl has become a defining characteristic of the Puget Sound lowlands (American Rivers 1999, Stoel 1999). In 1997, over three million people lived in the four county area surrounding Puget Sound, and another million are expected over the next 20 years (PSRC 1998; Figure 3). Urban growth, largely in the form of new suburban residential and commercial development, has outpaced even population growth. Between 1970 and 1990, the population of the Seattle metropolitan area increased by 38%, accompanied by a 87% increase in land development during the same time period (American Rivers 1999). Much of this urban development is concentrated along stream margins (Omernik and Gallant 1986).

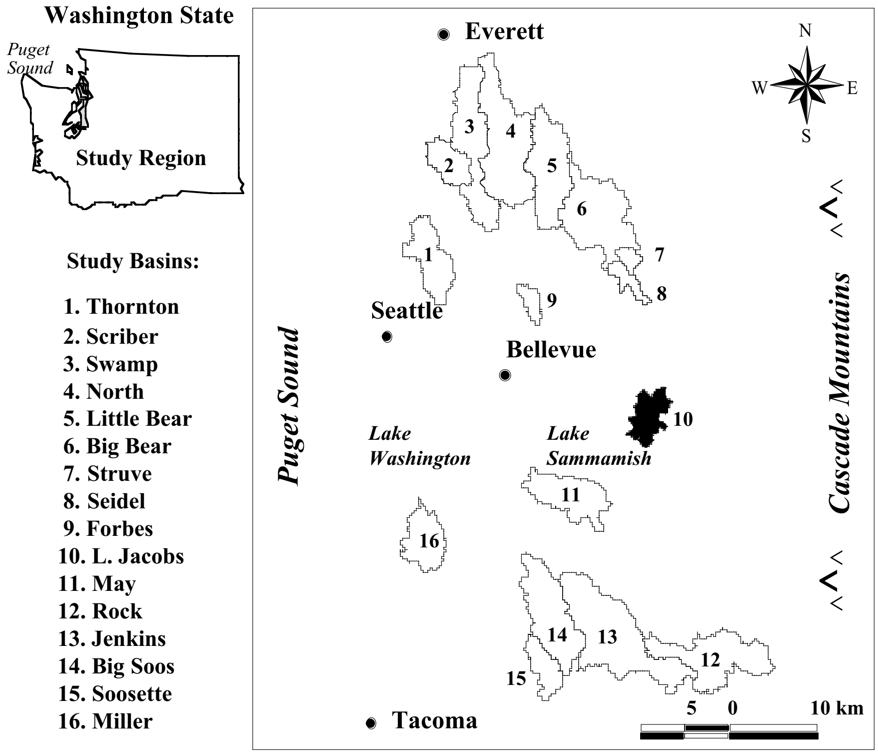

Figure 2. Map of study region. Location of study basins are shown relative to the four largest cities in the Puget Sound region (PSRC 1998). Note that basin boundaries are delineated for the most downstream sample site on each stream and that three basins (Scriber, Struve, and Seidel) are tributaries to other basins.

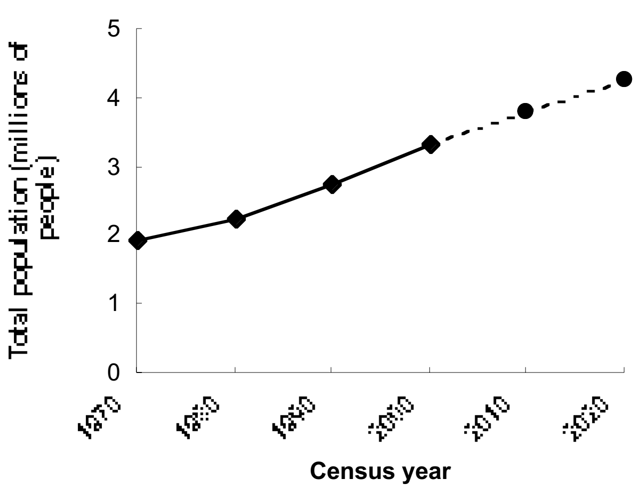

Figure 3. Population trends in the Puget Sound region (King, Kitsap, Pierce, and Snohomish counties.). Source: Puget Sound Regional Council (1998).

Site selection

Between 1997 and 1999, benthic invertebrate samples were collected from a total of 45 sites distributed among sixteen second- and third-order streams (Table 3, Appendix A). These streams are primarily located in the Lake Washington/Cedar River and Green River watersheds. Four of the 45 sites were sampled in more than one year. Across all sites for all years, drainage area ranged from 4 to 69 km^2 and elevation from 10 to 200m.

Table 3. Study basin area, land cover, and sampling intensity.

Study basins	Area (km^2) [1]	% Urban [1]	No. of sites
Lk. Washington / Cedar River			
Thornton Creek	25	91	4
Scriber Creek	15	84	1
Swamp Creek	58	70	10
North Creek	57	67	1
Little Bear Creek	40	54	9
Big Bear Creek	61	41	5
Struve Creek	4	48	1
Seidel Creek	7	19	1
Forbes Creek	5	85	2
Laughing Jacobs Creek	16	59	4
May Creek	30	36	1
Rock Creek	43	22	1
Green River			
Jenkins Creek	69	43	1
Big Soos Creek	42	61	1
Soosette Creek	14	63	2
Puget Sound			
Miller Creek	22	85	1

[1] values correspond to sample site furthest downstream

Land cover analysis. To examine the relationship between B-IBI and urban and forested land cover, 34 study sites were selected along a gradient of urban development. At the two extremes of land cover were Rock Creek in rural Southeast King county, and Thornton Creek within the Seattle city limits. The Rock Creek basin is covered primarily in second growth conifer forest and has much of its riparian corridor intact. This creek has been designated a “Regionally Significant Resource Area” (King County 1993).

Thornton Creek, one of the first basins in the Puget Sound region to be developed, is now one of the most heavily urbanized. The headwaters of this creek drain one of the oldest shopping malls in the nation. Fourteen of the 16 study basins typically had one or two B-IBI monitoring sites. Two basins (Little Bear and Swamp Creek) were sampled at nine and eight sites respectively (Figure 4). Multiple sites were selected to examine variation in biological condition within these heterogeneous basins.

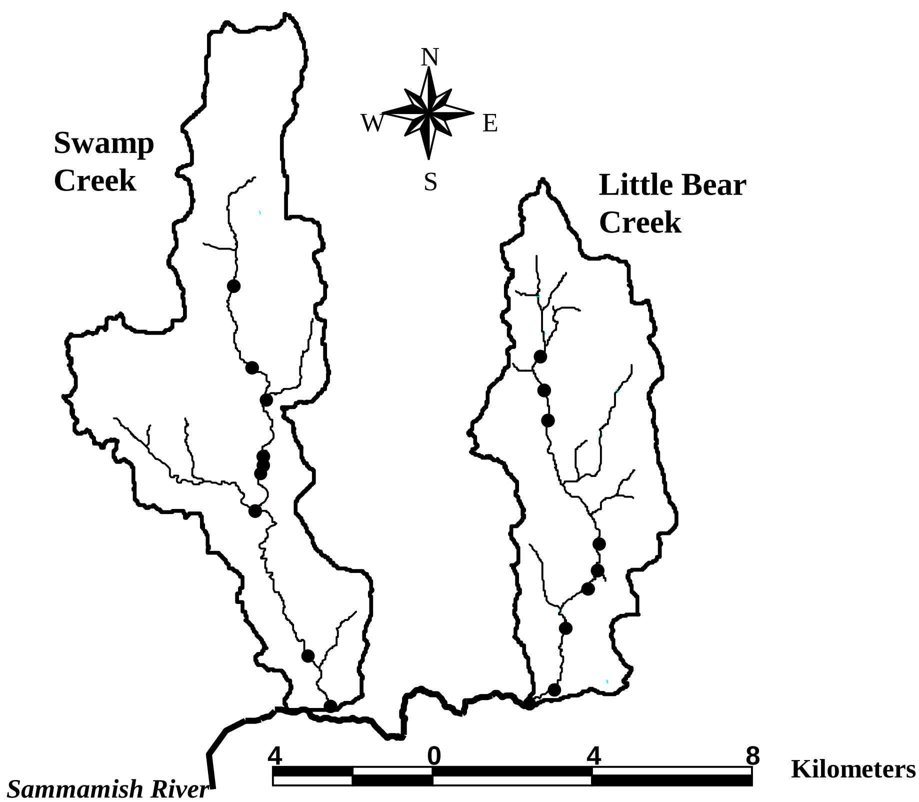

Figure 4. Distribution of study sites (●) along Swamp and Little Bear Creek. These two basins are in close proximity to each other and are of similar size, gradient, and geology. They differ primarily in the extent and pattern of urbanization relative to the stream channel. Note that only sites on the main channel of these streams were considered for within basin analysis and that sites immediately below restoration projects were excluded from land cover analysis.

Diagnostic evaluation. The relationships between B-IBI and substrate and flow features were tested across a sub-set of sites selected for land cover analysis. Substrate data were provided by a concurrent study at 18 invertebrate monitoring sites in 1997 (C.P.

Konrad, Department of Civil and Environmental Engineering, University of Washington, pers. comm. 2000). Hydrologic analysis was limited to monitoring sites located in close proximity to gauging stations without intervening tributary input. Hydrologic data for a sufficient period of record were only available at five invertebrate sites from this study. To increase sample size, six invertebrate sampling sites from an earlier study were included in the hydrologic analysis (Karr and Chu 1999a, Kleindl 1995). In total, 11 sites were analyzed for flow data from 1994-1995 and 1997-1998.

Restoration project assessment. Restoration efforts at five King County streams were selected to evaluate the response of invertebrates to LWD placement (Table 4). All five projects had been in place for at least a year prior to invertebrate collection, exceeded 200m in length, were located in moderate to highly urbanized basins, and listed salmon habitat enhancement as a project objective. B-IBI assessment was conducted in collaboration with concurrent evaluation of physical project condition (Larson 1999). Pre-construction invertebrate data were available for only one project (Soosette Creek; Greenberg 1995). In order to determine if the projects were successful in improving biological condition, monitoring sites were located immediately upstream (control) and downstream (treatment) of each restoration project. These paired sites were selected to be as similar as possible in all regards except for the placement of wood. At three of the projects, an additional mid-stream site was sampled between the control and treatment sites to test for localized effects. All five projects were sampled in 1998 and three of the more recently completed projects (Thornton, Swamp, and Laughing Jacobs Creek) re-sampled in 1999.

Sampling methods

Benthic macroinvertebrates. Invertebrates were collected from each site in September when flows are typically stable, taxa richness high, and field crews have easy access to sites (Fore et al. 1996). At each site, a Surber sampler (500-μm mesh, 0.1 m^2

Table 4. Restoration project and basin characteristics, ordered from most to least urbanized.

Stream:	Thornton	Forbes	Swamp	Soosette	L. Jacobs
Basin characteristics					
Land cover (% urban)	91	85	70	63	59
Drainage area (km^2)	25	5	53	14	16
Project Charactersitics					
Year constructed	1997	1988	1997	1994	1995
Project length (m)	280	210	370	1430	430
Project objectives					
Flood control	X	X	X		X
Sediment & erosion control	X	X		X	X
Habitat enhancement	X	X	X	X	X

project charactersitics and objectives from Larson (1999)

frame) was used to collect three samples along the mid-line of a single riffle. In the field, each sample was strained through a 500 -μm soil sieve, mineral material picked through and discarded, and the remaining sample preserved in a solution of 70% ethanol. Under microscopy, invertebrates were separated from remaining mineral and organic debris, identified, and counted. In this manner, each sample was processed and identified separately without compositing or sub-sampling (Doberstein et al. *in press*, Karr and Chu 1999a, Kerans and Karr 1994). Insect nymphs and larvae, the bulk of benthic samples, were identified to genus where practical (exceptions: Capnidae, Ceratopogonidae, Chironomidae, Dolichopodidae, Dystiscidae, Leuctridae, Phoridae, and Sciomyzidae); non-insect taxonomic identification varied from family to phylum (Appendix B). Non-benthic invertebrates, pupae, and terrestrial adults were excluded from sample analysis. Across all study sites, total taxa richness was 97, average replicate abundance 930 individuals, and median abundance 720. Two regional experts (W. Bollman, Rhithron Biological Associates, pers. comm. 1998; R.W. Wisseman, Aquatic Biology Associates, pers. comm. 1998) confirmed a complete reference set of invertebrates identified throughout the study.

Substrate and flow measurements. Three substrate and four hydrologic stream features were evaluated in relation to biological condition (Table 5). Size distribution of

Table 5. Substrate and flow features evaluated in relationship to B-IBI and selected metrics. An "**X**" indicates where the relationship between physical features and a particular biological metric was evaluated via simple linear regression.

Feature	**n**	**Description**	**Biology**				
Surface substrate			B-IBI	total taxa	EPT	clingers	long-lived
D_{50}	18	**mean:** diameter at which 50% of pebbles are smaller (mm)	**X**	**X**	**X**	**X**	
D_{16}	18	**fines[1]:** diameter at which 16% of pebbles are smaller (mm)	**X**	**X**	**X**	**X**	
$D_{84/BFD}$	18	**roughness:** 84% pebble diameter divided by bankfull depth	**X**	**X**	**X**	**X**	
Flow regime							
% > MAF	11	**flashiness:** percentage of year mean annual flow exceeded	**X**	**X**			**X**
Q_{max} : $Q_{inst.}$(daily)	10	**flashiness:** max. daily flow divided by max. instantaneous flow	**X**	**X**			**X**
$Q_{inst.}$: D.A.	10	**peak flow:** max. instantaneous flow divided by drainage area	**X**	**X**			**X**
Q_{max} : $Q_{min.}$ (annual)	11	**peak flow:** max. daily flow divided by min. daily flow	**X**	**X**			**X**

[1] D_{16} was strongly correlated with proportion of sub-surface fines (<2mm) for a sub-set of sites from this study (n=7, R^2=.91, p < 0.001; Konrad pers. comm. 2000).

surface substrate was characterized by a Wolman pebble-count, a method which consists of measuring the length of the intermediate axis of 100 randomly selected particles (Wolman 1954). Three measures were calculated from this data: D_{16}, D_{50}, and D_{84} divided by bankfull depth (C.P. Konrad, Department of Civil and Environmental Engineering, University of Washington, pers. comm 2000). D_{16} and D_{50} are commonly reported measures of particle size distribution; D_{84}/BFD refers to relative roughness, one parameter used in calculations of bank shear stress (Gordon et al. 1992). Hydrologic data were downloaded from continuous recording hydrologic gauging stations, and provided by King County Hydrologic Information Center (daily flow records), and Snohomish County Surface Water Management (15-min. records). Two measures each of peak annual flow and hydrologic flashiness were calculated from these data. Peak flow was expressed as the maximum daily and maximum instantaneous flow. Both were corrected for basin size by two different methods. Hydrologic flashiness is more difficult to capture. The percentage of time above mean daily flow was used to estimate both the increase in magnitude and frequency of peak flows and the decrease in baseflows that often result from urbanization (Booth and Jackson 1997; C.P. Konrad, Dept. of Civil and Environmental Engineering, University of Washington, pers. comm. 2000). The ratio of daily maximum flow to maximum instantaneous flow was also calculated to approximate flashiness. All hydrologic measures were averaged for the two years of flow record immediately prior to invertebrate collection. This timespan exceeds the aquatic life-cycle of the vast majority of invertebrates collected in this study (Merritt and Cummins 1996).

Geographical information systems (GIS). The extent of urbanization in each study basin was calculated over three spatial scales: sub-basin, riparian, and local (Figure 5). The land cover layer used in this analysis was a supervised land cover classification of a 1998 LandSat TM image with a mapping resolution of 30m (Botsford et al. 1998). This image was classified into seven categories of land cover appropriate to urban watershed analysis and error-checked against 1998 digital orthophoto quarter quadrangles (Table 6). A combination of vector and raster geographical data (Appendix C) was used to delineate

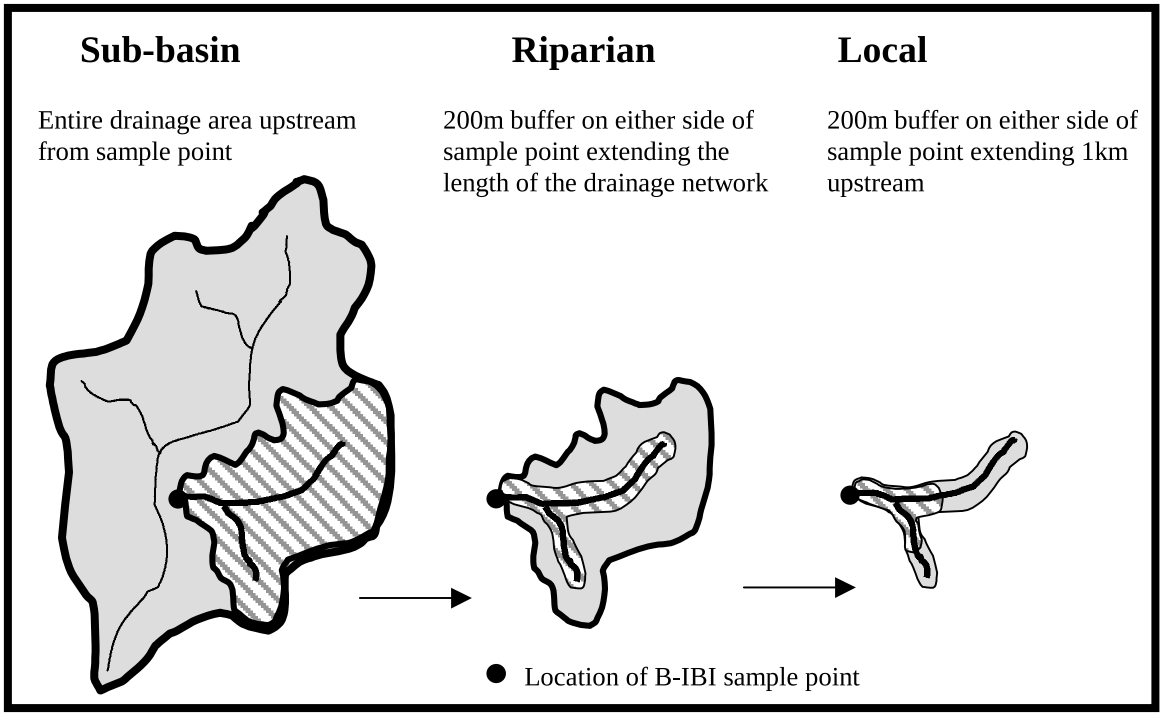

Figure 5. Diagram of GIS-based landscape analysis. Buffer widths dimensions were selected so as to be broad enough to include those functions commonly cited in association with riparian corridors (Gregory et al. 1991), but not unrealistically narrow given the relative accuracy of geographical datasets used in basin delineation and buffer analysis (Appendix C).

Table 6. Land cover categories defined from satellite classification. Values reported are from 1991 image (Botsford et al. 1998). 1998 accuracy assessment in progress (K.E. Hill, Department of Landscape Architecture, University of Washington, pers. comm. 2000).

	Actual land cover from orthophotos[1]			
Categories	open water	bare earth, pavement	grass, shrubs	trees
coniferous	0	1	8	91
deciduous	0	4	49	47
grass/shrub	0	29	63	8
water	100	0	0	0
urban (forested)	7	23	31	39
urban (grassy)	0	31	61	8
urban (intense)	9	62	21	8

[1] percentages, averaged for 100 pixels

unique sub-basins and buffers for each sample point. Within the GIS program ArcInfo, sub-basin delineation was based on flow direction and flow accumulation grids generated from a 10m-resolution digital elevation model. Within each of these sub-basins, two-hundred meter width stream buffers were selected with the buffer function in ArcView. Within each 200m buffer, the ArcView extension Network Analyst was used to trace 1000m reaches upstream of each sample point. The percentage of each land cover category in the contributing basin for each sample site was determined with map overlay functions for the three spatial scales.

Data analysis

Benthic index of biological integrity. Taxa richness, relative abundance, tolerance, and invertebrate life history information were used to calculate B-IBI scores for each study site. Following procedures first outlined for fish (Karr et al. 1986), and later for invertebrates (Fore et al. 1996, Kerans and Karr 1994, Kleindl 1995), raw scores from the three replicates were averaged for each of the 10 metrics—except for the "long-lived" and "intolerant" metrics which were calculated cumulatively. Based on deviation from reference condition and previously established scoring criteria for lowland streams of the Pacific Northwest, metric scores of one, three, or five were then assigned to the raw metric value (Table 7; Dewberry et al. 1999, Kleindl 1995, Fore et al. 1996). These 10 metrics scores were summed to provide a site and time specific B-IBI that ranges from 10 to 50.

Land cover. Four measures of land cover were tested as indicators of basin urbanization relative to B-IBI (Table 8). Two of these were composed of a single category of land cover and two were combinations of multiple categories. Graphical analysis and simple linear regression were used to evaluate the relationship between the four land-cover measures and B-IBI. This analysis was conducted over the three spatial scales for all study basins and separately within the Swamp and Little Bear basins.

Table 7. Scoring criteria for small (second-fourth order) lowland Pacific Northwest streams. A score of 5 indicates little or no deviation from reference condition; a score of 3 indicates moderate deviation; a score of 1 indicates strong deviation. Values that fall immediately on scoring break are assigned the higher score. Note that scoring criteria will differ depending on level of taxonomic identification.

B-IBI Metric	1	3	5
total taxa	< 14	14 - 28	> 28
mayfly taxa	< 3.5	3.5 - 7	> 7
stonefly taxa	< 2.7	2.7 - 5.3	> 5.3
caddisfly taxa	< 2.7	2.7 - 5.3	> 5.3
% dominance	> 75	55 - 75	< 55
long-lived taxa	< 4	4 - 8	> 8
intolerant taxa	< 2	2 - 4	> 4
% tolerant taxa	> 44	27 - 44	< 27
clinger taxa	< 8	8 - 16	> 16
% predators	< 4.5	4.5 - 9	> 9

Table 8. Land cover measures tested as measures of basin disturbance and their expected relationship to biological condition.

Attribute (% of area)	Land cover categories included in analysis	B-IBI response
conifer [1]	coniferous	positive
forested	coniferous + deciduous	positive
intense urban	intense urban	negative
urban	intense urban + forested urban + grassy urban	negative

[1] Coniferous vegetation is the native land cover for the region.

Diagnostic evaluation. Along with overall B-IBI, specific metrics were analyzed in relation to the seven physical stream channel features (Table 5). Clinger taxa richness (adapted for attachment to surfaces such as rocks) were evaluated relative to change in substrate features as was combined EPT taxa richness (Mayflies, Stoneflies, and Caddisflies). This grouping includes many of the more sensitive taxa found in streams, many of which are obligate erosional (found only in riffles—thus more likely to be susceptible to sedimentation; Richards et al. 1997). Long-lived taxa richness (merovoltine: requiring more than two years to complete life cycle or living more than two years) was examined in relation to hydrologic disturbance. Total taxa richness was evaluated relative to both hydrologic and substrate features. Graphical analysis and simple linear regression were used to analyze the relationships between B-IBI and metrics to substrate and flow.

Restoration project assessment. A paired t-test and its non-parametric equivalent, the Mann-Whitney U test (Zar 1996), were used to compare B-IBI (parametric) and metric (non-parametric) scores above and below restoration projects. Additionally, change in total invertebrate abundance above and below projects was evaluated with a Mann-Whitney U test. Although natural variation in abundance is typically very high (and thus is a poor indicator of stream condition; Karr and Chu 1999a: premise 25), similar studies have reported invertebrate abundance (with mixed results) as it relates to food availability for fish (Angermeier and Karr 1984, Hilderbrand et al. 1997).

RESULTS

B-IBI

Invertebrate biota of nearly all sites sampled in this study indicated mild to severe stream degradation. Although B-IBI varied from 10 (very poor) all the way up to 48 (excellent; Figure 6), only 10% of sites sampled across the study were comparable or only slightly divergent from reference condition for the region. The best biological conditions were observed at Rock Creek, where 44 taxa were found across the three replicates; eight of these were stoneflies, nine classified as long-lived, three as intolerant, and 11% of total individuals as predators. Many taxa, such as the stonefly genus *Pteronarcys* which can take up to four years to reach maturity (Stewart and Stark 1993), were found only at Rock Creek and one or two other similarly least disturbed sites. In contrast, Thornton creek had only 15 taxa, no stoneflies, one long-lived, and no intolerant taxa or predators. Amphipoda, chironomidae, and a tolerant mayfly made up 89% of total individuals across the three replicates at Thornton Creek. The biological condition at the rest of the sites from this study fell in between these two extremes. Along the multiple sites on Little

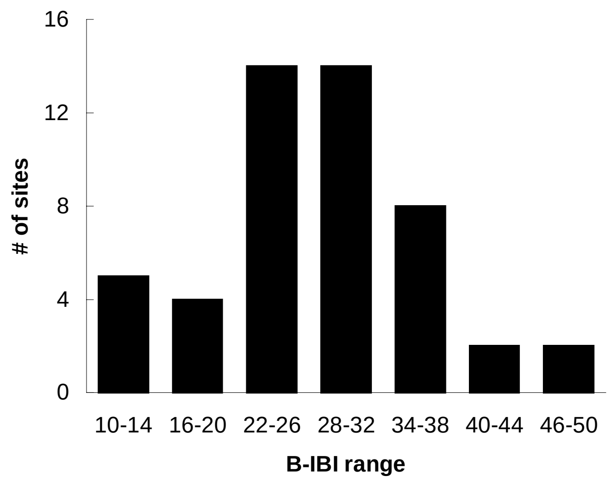

Figure 6. Distribution and range of B-IBI across all study sites (median = 28, mean = 27.4, standard deviation = 8.9; Appendix D).

Bear and Swamp Creek, B-IBI told two very different stories. On Little Bear, biological condition was good in the headwaters with a B-IBI of 40, but this score rapidly dropped down to 16 over a distance of approximately 10 km. In contrast, B-IBI varied relatively little between a high of 32 and a low of 22 along a 14 km. length of Swamp Creek.

Why is between site biological variability so different between these two very similar basins? Extending this question to all study sites, what are some of the primary factors controlling the range in biological condition observed across Puget Sound lowland streams? Neither basin size nor sample year explained significant variability in B-IBI across the streams in this study (Figures 7 and 8). Clearly, level of urban development was a key factor in the distribution of site scores: highest B-IBI's (>38) were concentrated in less developed headwaters and unincorporated areas of King and Snohomish counties, while scores of 16 or less were located in areas of high residential and/or commercial development. This relationship is captured only crudely by impervious area. B-IBI along sites in lower Little Bear Creek for example are again much lower than predicted by this measure (Figure 9). Does a broader definition of urbanization derived from the land cover classification provide a better explanation of B-IBI pattern?

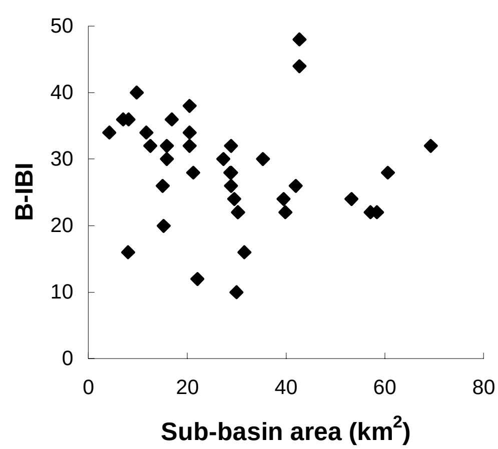

Figure 7. The relationship between B-IBI and sub-basin area (R^2 = 0.02, p > 0.10).

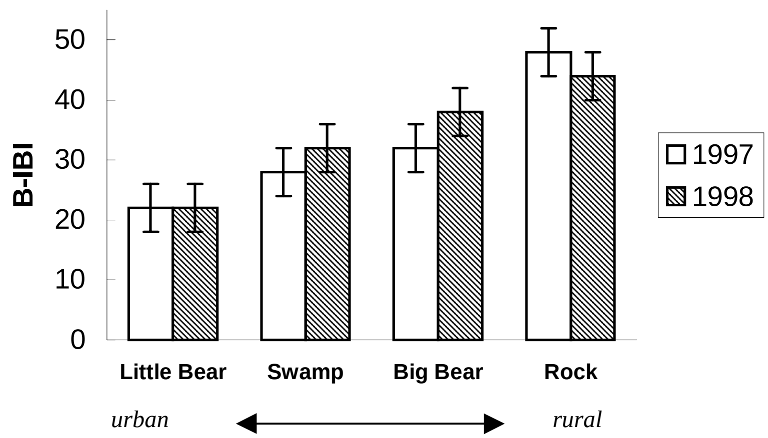

Figure 8. Variation in B-IBI between sample years. Values are plotted for the four sites sampled in both 1997 and 1998 (n=2). Generally, B-IBI scores must differ by at least four points to conclude that sites are significantly different (Doberstein et al. *in press*).

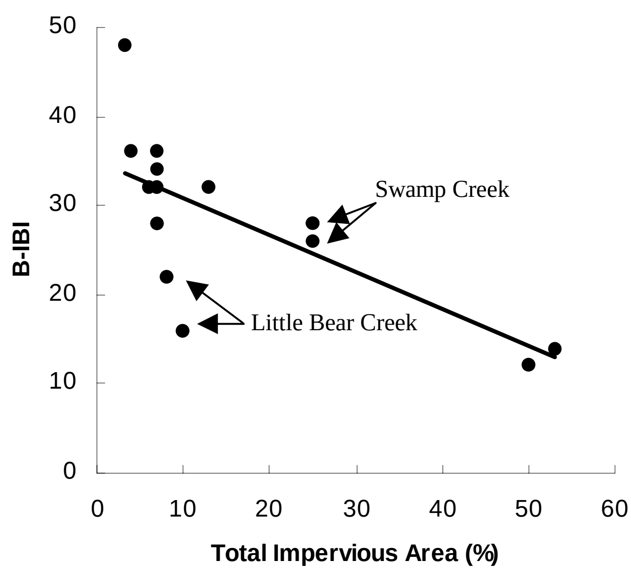

Figure 9. The relationship between B-IBI and total impervious area (1997). Values are plotted for the 16 sites at which comparable impervious area data were available (Source: Kleindl 1995, May 1996).

Land cover

The distribution of land cover among the 16 basins of this study reflects current development trends around the Puget Sound lowlands—conversion of forested lands to urban and suburban centers (Figure 10). The 34 study sites located within these sixteen study basins followed a normal distribution along the forested to urban gradient, with the greatest number of sites near a 50/50 split between urban and forested categories (Figure 11). Not surprisingly then, two proposed land cover measures (forest and urban land cover) were near perfect inverse measures ($R^2 = 0.98$, $p < 0.001$). To avoid redundancy, forest cover was discarded from further analysis among spatial scales and with B-IBI.

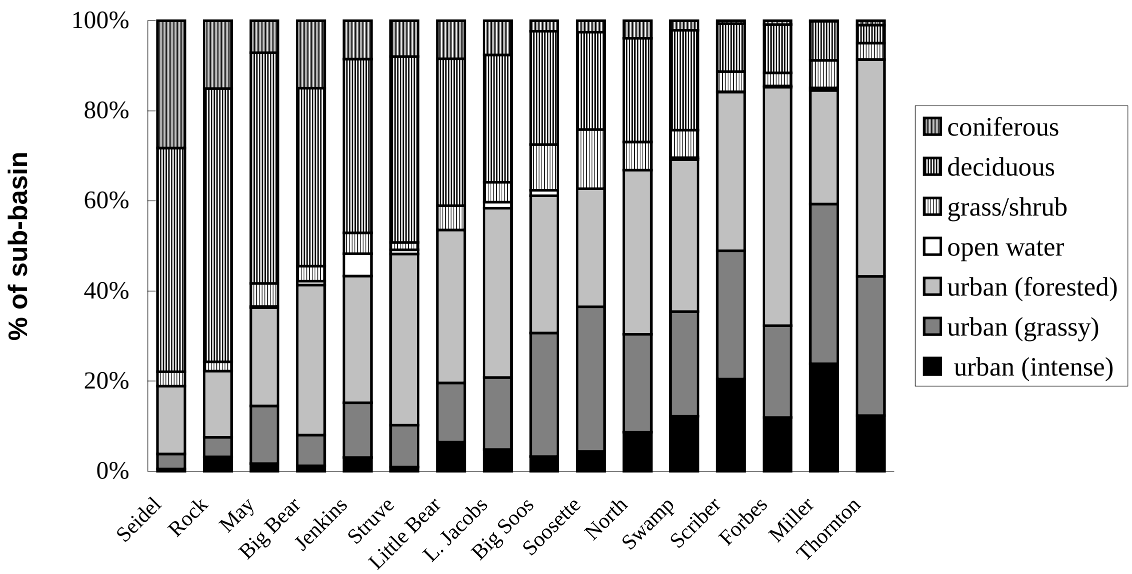

Figure 10. Distribution of land cover categories within study basins. Basins are ordered from least to most urban. With the exception of open water, all seven land cover categories are present in each of the 16 study basins. Combined forested categories range from 5-78% of the total area of the basin, and combined urban categories from 19-91%.

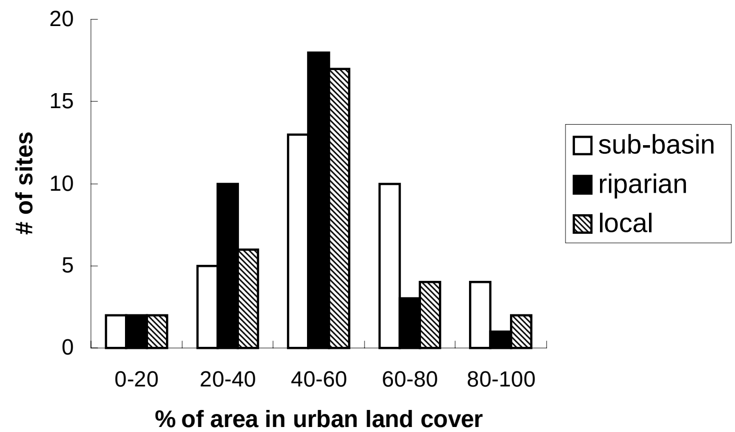

Figure 11. Distribution of study sites across a gradient of urbanization.

Of the remaining three land cover measures tested, the combination of urban land cover categories was best correlated with B-IBI (Table 9), and is used throughout this study in relation to biological and physical stream response. This measure of urbanization, however, was not independent among scales. Both the riparian and local land cover scales defined in this study varied as a function of land cover at the broader sub-basin scale (Figure 12). This is not an unexpected result, particularly considering the relatively wide width used to define the riparian and local scale. Because riparian and sub-basin land cover were so closely correlated (R^2 = .95, p < 0.001), the remainder of this study focuses on B-IBI response at the sub-basin and local scales.

Table 9. Summary table of the relationships of biology to land cover. R^2 values are reported for linear regression between B-IBI and three measures of basin development over three spatial scales: sub-basin (n=34), riparian (n=34), and local (n=31).

R^2	Land cover measure		
Spatial scale	**conifer**	**intense urban**	**all urban**
sub-basin	0.35	0.34	0.53
riparian	0.34	0.45	0.56
local	0.49	0.17 *	0.49

* p < 0.05, all other values < 0.001

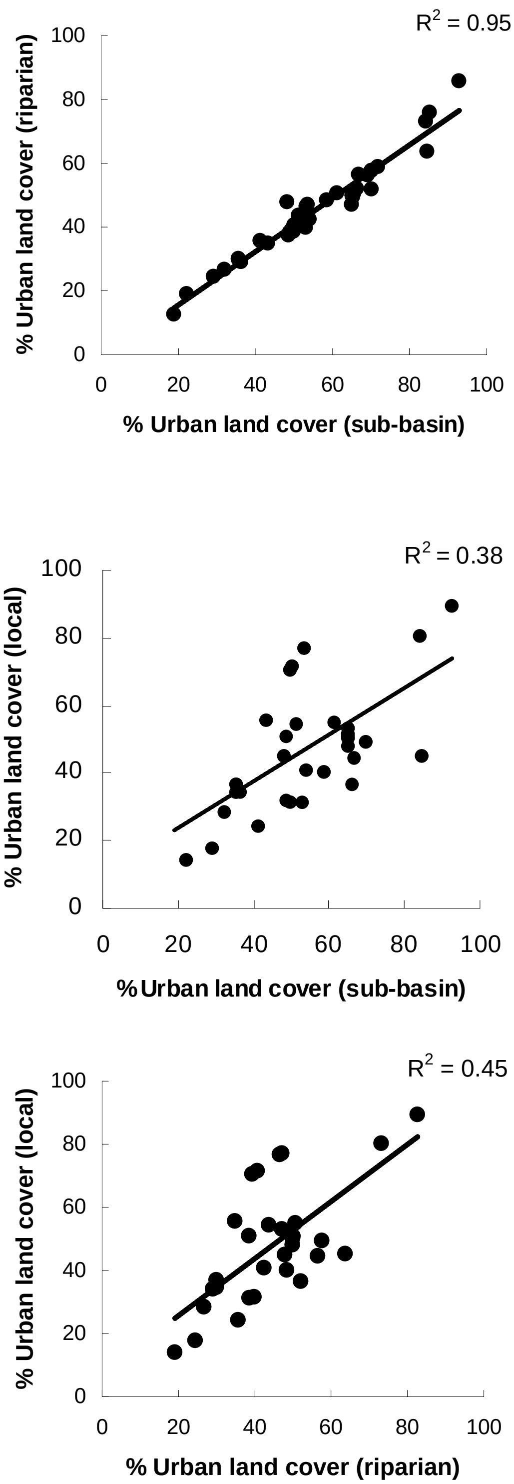

Figure 12. Correlation in urban land cover between spatial scales ($p < 0.001$).

The spatial pattern of development within the intensely sampled Swamp and Little Bear Creek basins differed in several important ways. At the sub-basin scale, Swamp Creek was *more* urbanized than Little Bear, with 70% vs. 54% urban land cover, respectively. But at the local scale, the reverse pattern was observed: all sample sites on Swamp Creek were *less* urbanized than the six sites on lower Little Bear. Although more developed overall, Swamp Creek had a more forested riparian corridor upstream of sampling sites than did Little Bear.

B-IBI *v* land cover

How does this contrasting pattern of development relate to the biological condition observed across sites in these two basins? In Little Bear Creek, B-IBI variability was strongly related to local land cover change (Figure 13a). The maximum B-IBI (40) on this stream occurred at the site with the least amount of local urban land cover (32%) in comparison to the low (16) with 71% local urban land cover. Interestingly, this score of 16 increased to 30 after the stream entered a reach with a more forested riparian corridor (local urban land cover = 54%), but then dropped back down to 22 after resuming course through the City of Woodinville (local urban land cover = 77%). In contrast, extent of sub-basin urban land cover varied only between 49 and 54% across the nine study points on this stream. When partial sub-basin land cover was calculated (i.e., portion of sub-basin above sample point but below next upstream sample site), urban land cover varied between 33 and 91%. Neither of these sub-basin scales, however, explained a great deal of variability in B-IBI. In Swamp Creek, neither sub-basin nor local urban land cover varied substantially (Figure 13b), an observation that is concordant with limited variability in B-IBI.

Can the patterns observed in these two basins be generalized across the entire study region? Focusing primarily on within basin land cover changes may bias results toward detecting primarily smaller-scale effects (Allan et al. 1997). When the relationship

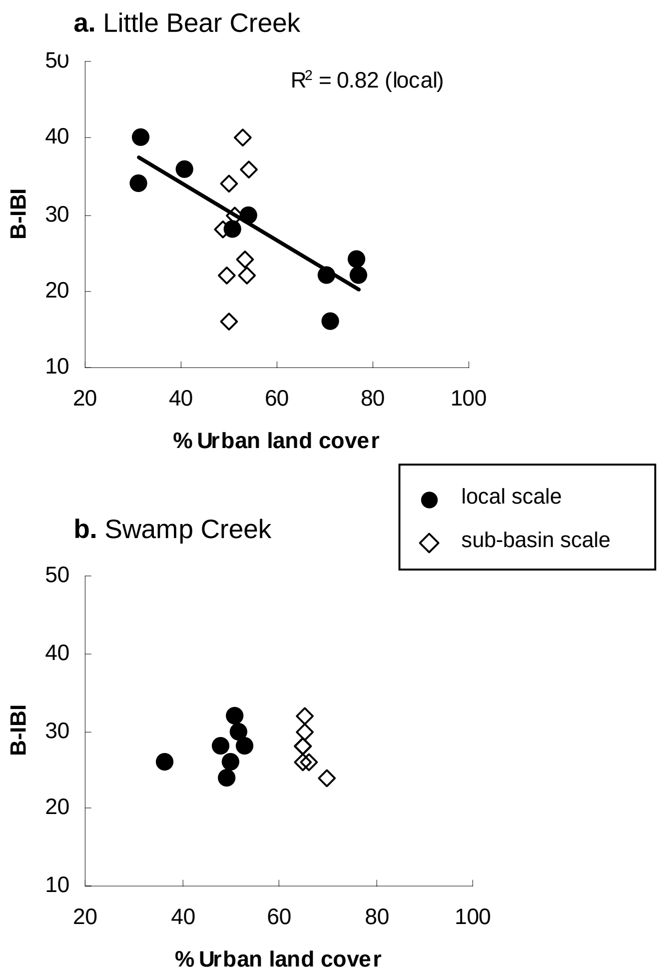

Figure 13. B-IBI *v* urban land cover in Little Bear and Swamp Creek. Note that for Swamp Creek, the most downstream site is excluded; local land cover could not be accurately determined here due to discrepancies between geographical datasets.

of B-IBI to urban land cover is examined across the 34 sites in all 16 study basins, sub-basin and local land cover explained approximately equal amounts of variability in B-IBI (Figure 14). Clearly, the biological condition of a stream is inversely related to the extent of urban land cover in its basin. This relationship holds at both the local and sub-basin scale of analysis tested here. When these two scales are combined in a multiple regression model, urban land cover explains 59% of variability in B-IBI ($p < 0.001$). Both of these scales interact to strongly affect biological condition, but land cover change alone does not capture every type of disturbance that affects streams. Does an analysis of flow and substrate conditions shed any more light on between site biological variability?

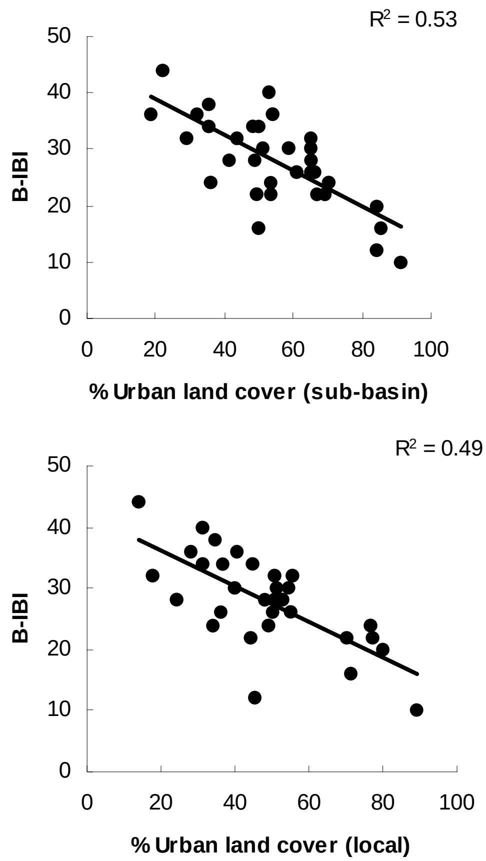

Figure 14. B-IBI *v* urban land cover across all sites.

B-IBI *v* substrate and flow

Pattern in B-IBI and metrics across sites was explained to some degree by local channel features. Total index values were better related to these features than any of the selected metrics. Of the seven channel measures tested, three (RR, % > MAF, and Q_{max}:Q_{inst}) were statistically related to biological response and/or to the extent of upstream urbanization (Table 10). Relative roughness was one such measure, although graphical analysis reveals that the strength of this relationship relies heavily on one sample point (Rock Creek; Figure 15). The two particle size distribution measures (D_{16} and D_{50}) were poorly related to B-IBI. Both were better predicted by sub-basin area than urban land cover, suggesting that variation in particle size was more a factor of natural basin differences than anthropogenic impacts. Alternatively, lack of correlation between particle size and urbanization may also reflect channel armoring practices; placement of rip-rap in urban stream channels artificially increases substrate size distribution. Of the flow features, both measures of hydrologic flashiness were correlated with biological response and with sub-basin area, particularly the percentage of time above mean annual flow (Figure 16). Although not statistically significant, hydrologic flashiness was better related to urbanization at the sub-basin rather than local scale. In contrast, neither measure of peak flow explained any degree of variability in B-IBI or metrics; invertebrates seem to respond more to the degree of flow fluctuation than to the magnitude of peak events.

Table 10. Summary table of substrate and flow features.

R^2	Substrate			Flow regime			
	fines	median	roughness	flashiness		peak flow	
	D_{16}	D_{50}	D_{84}:BFD	% > MAF	Q_{max}:$Q_{inst.}$	$Q_{inst.}$:D.A.	Q_{max}:$Q_{min.}$
Sub-basin area	0.18 [1]	0.09	0.05	0.50 [2]	0.45 [2]	0.18	0.00
Urbanization							
% urban (basin)	0.00	0.01	0.28 [2]	0.38	0.08	0.00	0.01
% urban (local)	0.02	0.05	0.49 [2]	0.00	0.03	0.00	0.11
Biology							
B-IBI	0.08	0.03	0.39 [2]	0.67 [2]	0.32 [1]	0.25	0.00
total taxa richness	0.09	0.03	0.21 [1]	0.39 [2]	0.31 [1]	0.03	0.07
EPT richness	0.15	0.04	0.16 [1]	—	—	—	—
clingers richness	0.15	0.04	0.18 [1]	—	—	—	—
long-lived richness	—	—	—	0.33 [1]	0.11	0.00	0.03

[1] $p < 0.10$, [2] $p < 0.05$

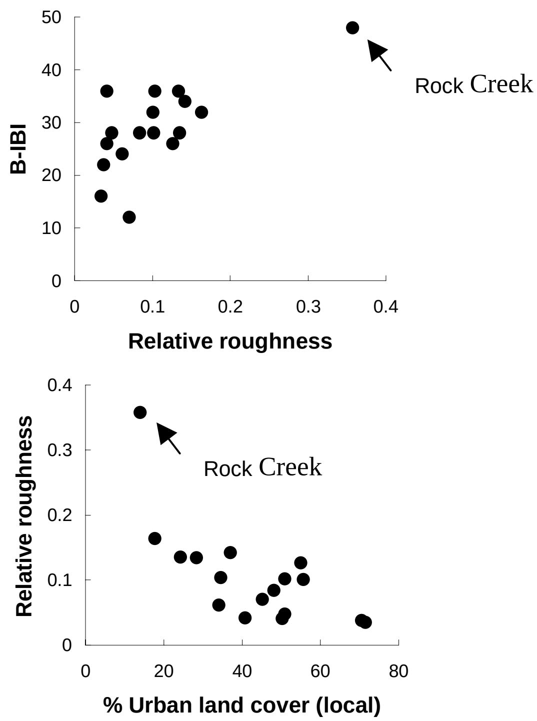

Figure 15. Relationship of relative roughness to B-IBI and urbanization.

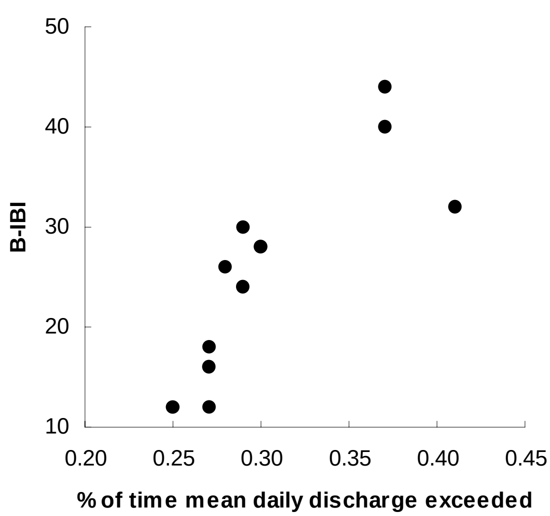

Figure 16. The relationship between B-IBI and hydrologic flashiness (R^2 = 0.67, p < 0.01).

Restoration project evaluation

Addition of LWD had little demonstrable effect on biological condition as measured by B-IBI (Figure 17). This finding parallels results from the physical evaluation at project sites. Although pool spacing did decrease as a function of increased LWD, added wood generally did not locally improve sediment retention or reduce bank erosion (Larson 1999). Two projects (Thornton and Swamp Creek) had been in place only one year when this study was conducted and although invertebrates rapidly recolonize the benthos following disturbance, this process may take longer if sources of colonizers are more distant (Gore 1985), or if the channel is still equilibrating (Booth et al. 1997). Additional sampling in 1999, however, still showed no improvement in biological condition at Thornton, Swamp, or Laughing Jacobs Creek. Nor was there an improvement when samples were collected within project boundaries (as opposed to immediately downstream). Overall invertebrate abundance between control and treatment sites was also not significantly different (Mann Whitney U-test, $p > 0.10$). Of the ten metrics, total taxa richness, Trichoptera taxa richness, and dominance significantly improve downstream of the five projects (Mann Whitney U-test, $p < 0.10$). These findings depend largely on the differences observed between the control and treatment sites at one project—Soosette Creek.

Overall, the B-IBI scores at all projects evaluated for this study were much better correlated with the level of local urban land cover than with the presence or absence of a LWD project. Generally, B-IBI scores must differ by at least 4 points (Doberstein et al. 2000) to conclude that sites are significantly different. Post-treatment B-IBI on Soosette Creek scored significantly higher than for either the upstream control site or pre-project collection. Local land cover analysis reveals that the “control” site on this creek was considerably more urbanized than the treatment (53% vs. 13%), and thus serves as a poor comparison by which to judge the effects of LWD addition. Drawing comparisons

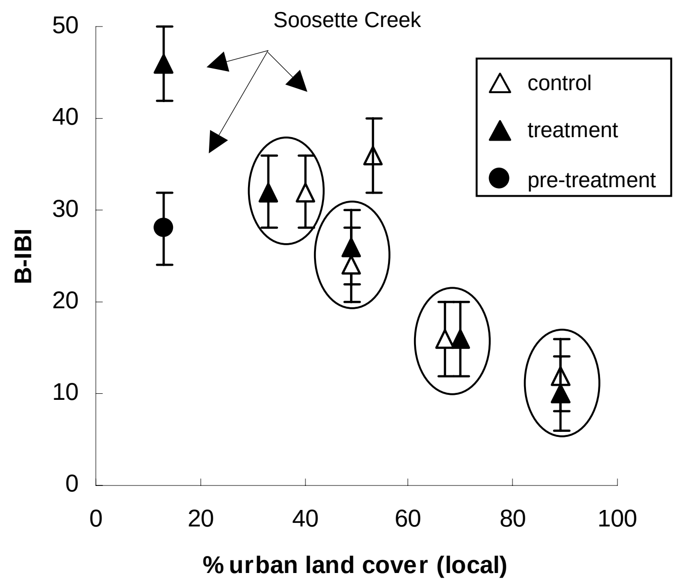

Figure 17. Paired B-IBI scores for restoration monitoring. Overall, control and treatment B-IBI at the five projects were not significantly different (paired t-test, $p > 0.10$). Soosette Creek is the exception.

between pre- and post-treatment condition is also problematic, as sampling methodology differed slightly between years. Furthermore, the installation of wood on Soosette occurred in 1993 as mitigation for a road failure and massive debris flow that occurred during winter, 1990 (Greenberg 1995). It is difficult to know whether the biological improvement observed between sampling years is due to restoration activities or to the process of natural downstream recovery over the last decade.

DISCUSSION

Biological condition of Puget Sound lowland streams

Extensive and diverse activity throughout the Puget Sound basin has altered the region's landscapes—with especially devastating effects on stream biota. Half of the stream sites sampled in this study were in poor biological condition; almost all sites lacked even a single intolerant taxon and at the most urbanized sites no stoneflies were found. Although the sites from this study were not randomly selected, such degraded conditions are typical of many streams in and around major metropolitan areas in the region (Fore et al. *in press,* Karr and Chu 1999a, Kleindl 1995). Given the rate at which remaining forested areas around the lowlands are being replaced by urban and suburban centers, there is significant cause for concern. The survival of wild salmon in the Pacific Northwest depends on many factors, crucial among them being high quality streams for spawning and rearing of young. In order to protect such healthy streams that remain, and restore those that have been degraded, it is essential to focus on biology. This refers not only to salmon production, but to the overall health of the streams and rivers upon which these fish depend. As one such measure of stream biological condition, B-IBI has broad application to both the design and evaluation of restoration efforts. In this study, B-IBI was combined with analysis of channel and landscape condition to address questions concerning urban land cover, spatial scale, diagnostic properties of B-IBI, and the biological effects of LWD placement.

Measuring urbanization—beyond imperviousness. Urban development degrades streams (Lenat and Crawford 1994, Roesner 1997, Steedman 1988). But is it possible to more precisely describe the relationship between urbanization and biological condition, and thus to make better urban stream management and restoration decisions? None of the measures of land cover tested in this study were perfect fits with B-IBI, but overall, a grouping of equally weighted urban land cover categories explained a high degree of variability in B-IBI. This simple yet broad definition of urbanization is more inclusive of a variety of potential impact types than what is captured by impervious area models. Even in areas of the urban basin that aren't paved over, compacted soils rarely retain the high infiltration rates associated with forested areas and reach saturation more rapidly with increased runoff from adjoining paved surfaces (Dunne and Leopold 1978). Going beyond changes in flow regime, a residential yard adjacent to a stream may not contribute the same volume of surface runoff as does a paved surface, but it may still substantially alter the quality of that runoff. A grassy lawn also provides few, if any, of the benefits associated with native riparian vegetation. Because humans modify watersheds in many ways, a broad definition of anthropogenic disturbance is appropriate for use in conjunction with biological assessment. Similarly, basin management efforts should focus not only on limiting impervious surfaces, but on retaining forested cover, protecting wetlands and riparian corridors, and other details in the multiple dimensions of stream degradation**.**

The importance of spatial scale. Taking a broader definition of disturbance refers also to examining how urban development influences stream condition over multiple spatial scales. B-IBI in the urban streams of this study responded strongly to land cover change over both the entire sub-basin and local scale. The relative predictive strength of one scale over another depended largely on whether sites were located within or across stream basins. A growing body of research has illustrated the importance of basin-wide land cover features in shaping the biological and physical features of streams (Richards et al. 1996, Roth et al. 1996, Wang et al. 1997). Results from this study of urban Pacific Northwest streams support these findings, but indicate that local scale changes in land cover are also ecologically significant (Barton et al. 1985, Scarsbrook and Halliday 1999, Steedman 1988). The geomorphic and biochemical attributes of streams are themselves controlled at different spatial scales (Frissell et al. 1986) and any of these may in turn potentially limit biological condition (Allan 1995). Rarely is land cover homogenous across urbanizing basins (Wear et al. 1998) and as a consequence biological condition may also vary substantially along a length of stream. Sampling at only one site along an urbanizing stream will rarely give a complete picture of overall stream condition. Similarly, measuring land cover at any single spatial scale does not capture the full range of ecological impacts that urban development has on streams. As with chemical and physical parameters, no single measure of basin development is adequate as a surrogate for directly monitoring biological condition and used as such will greatly underestimate extent of stream degradation.

Diagnostic properties of B-IBI. B-IBI responded predictably across a gradient of urbanization, but it was also sensitive to changes in substrate and flow conditions. In particular, channel roughness and hydrologic flashiness were both correlated with B-IBI. High values of relative roughness, as observed on Rock Creek, may indicate a greater diversity of flow conditions (e.g., availability of slow-water refugia) during high flow events (Borchart 1993, Davis and Barmuta 1989). In terms of flow regime, Rock Creek was also one of the least flashy sites. Stream invertebrates are adapted for life in strong currents, but few are able to exist under conditions of extreme and unpredictable flow fluctuation (Borchart and Statzner 1990, Irvine 1985, Poff and Ward 1989). The two most urban basins in this analysis (Miller and Kelsey) were also the flashiest; biological condition at sites on both of these creeks was severely degraded.

Invertebrates are affected by diverse stream features—both natural and anthropogenic—that interact with and potentially mask the limited set of parameters tested here. At extreme expressions, flow and/or substrate features may limit biological condition. At more moderate values other limiting factors likely come into play. For instance, the influence of particle size varies as a function of current, food availability, species interactions, and life stage (Culp et al. 1983, Minshall 1984, Williams 1980), not to mention sediment mobility and subsurface characteristics. No single physical, chemical, or hydrologic measure alone can accurately predict biological condition in all cases. Unfortunately, this has often been the traditional approach by which estimates of the health of aquatic biota have been made. Instead of using physical or hydrologic measures as surrogates for biological condition (or vive-versa), further study of the relationships between B-IBI and metrics to diverse urban impact types (physical, hydrologic, chemical, etc.) is needed. Although the results from this component of the study were not entirely conclusive, diagnostic evaluation is an important direction for the field of biological assessment—particularly as it relates to evaluation of stream restoration efforts.

Are current models of restoration working? Overall, B-IBI did not detect any substantial positive effect on biological condition from the restoration activities at the time scales sampled. Only one of the six projects evaluated exhibited higher B-IBI, and here study design limitations prevent attributing this to LWD addition alone. Physically, the effectiveness of LWD in urban streams may be substantially limited by severe modification in channel form and flow regime (Booth et al. 1997). Much of the wood installed in these five projects was undersized for the stream and consequently washed out during high flow conditions, or in some cases was stranded above incising channels (Larson 1999). Biologically, placing logs devoid of bark, roots, branches, or leaves into urban streams is not equivalent to natural recruitment, where wood is but one benefit of a forested riparian corridor and comes in a variety of forms, sizes, and configurations (Bilby and Ward 1989, Gregory et al. 1991). Adding wood to streams is not necessarily misplaced, but taken alone this activity does not address the more important issue of why wood is lacking from urban streams in the first place, or what else is amiss. As illustrated by the diagnostic component of this study, many types of urban stressors affect the aquatic biota simultaneously. In order to achieve meaningful long-term biological recovery, restoration efforts must take a broad focus. This entails looking beyond narrow conceptions of local scale in-stream habitat manipulation to address additional local impacts and factors operating across the entire basin (Ziemer 1997).

Management applications

Setting restoration goals. What can localized restoration projects realistically achieve in biological terms? Results of the land cover analysis indicate that the success of localized efforts in improving biological condition will likely be limited by extent of sub-basin urbanization. The combined biological and land cover analysis presented in this study, therefore, provides a useful context in which to set restoration goals.

Understanding why some sites score higher than predicted by land cover measures can also serve to guide future development in ways that minimize effects to stream biology (e.g., leaving wide and contiguous riparian corridors intact). At sites with lower B-IBI than predicted, more detailed assessment is appropriate to determine what is limiting biological condition, and how those specific impacts can be addressed by management and restoration actions. For example, the City of Woodinville, through which Little Bear Creek runs, is currently working to acquire and protect remaining patches of forested corridor (D. Knight, Executive Branch, City of Woodinville, pers. comm. 2000).

Riparian corridor conservation. The role of forested riparian corridors in protecting stream health in human modified landscapes has been described both in terms of benefits conferred: stream shading, bank stabilization, and inputs of leaf and wood debris (Gregory et al. 1991, Naiman et al. 1993, Sweeney 1993) and in terms of the deleterious impacts corridors "buffer": the erosive hydrologic force of peak runoff events, high nutrient and sediment loads, and trampling by both humans and livestock (Karr and Schlosser 1978, Osborne and Kovacic 1993, Schlosser 1991). The effectiveness of corridors in achieving these functions and thus protecting biological condition will depend largely on corridor dimensions (laterally *and* longitudinally) and the extent of upland development (FISRWG 1998). Recent studies in forested and agricultural basins indicate that biological communities may show signs of recovery even over relatively short (< 300m) lengths of forested corridor (Scarsbrook and Halliday 1999). The corollary has also been demonstrated; in largely undisturbed basins, local patches of deforested riparian corridor are still associated with deleterious impacts to aquatic taxa (Jones et al. 1999). How do these findings apply in predominately urban landscapes where forested corridors, if they exist at all, are typically both narrow and discontinuous?

My results indicate that the effectiveness of localized patches of riparian corridor in maintaining biological integrity varies as a function of basin-wide urbanization. In Little Bear Creek, high B-IBI was associated with sites located in headwater reaches of intact riparian corridor. Further downstream, B-IBI decreased dramatically as local riparian vegetation was replaced by roads, houses, and commercial centers. B-IBI did increase significantly through a local reach of forested corridor, although this increase was temporary and still not as high as in the headwaters of the creek. In short, when overall basin development is low to moderate, forested riparian corridors have significant potential to positively influence biological condition. At the same time, even small patches of urban land conversion in riparian areas can still severely degrade stream biology. Neighboring Swamp Creek was more urbanized at a sub-basin scale but less so along the stream margin. Although B-IBI throughout this stream never indicated the severe degradation observed on lower Little Bear, sites on Swamp creek also never scored particularly high. In some of the most urban basins of this study (e.g., Forbes and Miller Creek), B-IBI was still very poor even in reaches with some degree of forested corridor. In such highly urban basins, severe hydrologic alteration may potentially overwhelm local corridor benefits, while for a stream such as Little Bear, local channelization and loss of riparian vegetation may be more limiting factors on biological condition. As both a conservation and restoration strategy, protection and reforestation of riparian areas is critical for preventing severe stream degradation (Osborne et al. 1993), but alone these measures are not adequate to maintain biological integrity in streams draining highly urban basins (Roth et al. 1996).

Testing biological attributes. Having a robust measure of basin urbanization is also important for validation and testing of future evolutions of biological indexes. Each of the current ten metrics of the B-IBI (with the exception noted below) responds predictably along a gradient of urban land cover (Figure 18). This dose-response relationship between biology and anthropogenic disturbance is one of the critical underpinnings of multimetric indexes (Karr and Chu 1999a). The intolerant taxa richness metric was the one biological attribute not well correlated with sub-basin urbanization in this study—likely reflecting in part the extent of disturbance in the region. Across all study sites for all sample years, only six taxa classified as intolerant were identified, and these in extremely low abundance. Most intolerant taxa were completely absent from the majority of sites in this study (Figure 18g). When tested across a less developed region, the intolerant taxa metric responds in a more predictable fashion (Dewberry et al. 1999, Karr and Chu 1999a). Another factor that may confound the utility of this metric is in the definition of intolerance. As used in this study, the term refers to tolerance to disturbance in the general sense rather than to sensitivity to a specific impact type (e.g., heavy metals or organic enrichment; Wisseman 1999). As more detailed life history and tolerance information is learned about the indicator organisms used in biological assessment, indexes such as the B-IBI will need to evolve along with this knowledge. For these reasons, it is particularly important to have a replicable measure of disturbance (such as sub-basin or local urbanization) with which to fine tune our understanding of the best biological measures of stream condition and their relation to human influence.

Future study directions

The land cover analysis presented in this study has many potential management applications, but in and of itself is not an indicator of stream condition. Any method of measuring anthropogenic disturbance is at best an imperfect predictor of stream biological condition. Satellite imagery, though very useful for looking at land cover patterns across a region, indicates little about the nature of the constructed drainage network, the extent of stream channelization, the quality of urban stormwater runoff, or the presence of exotic species. To some extent, these features vary predictably along the

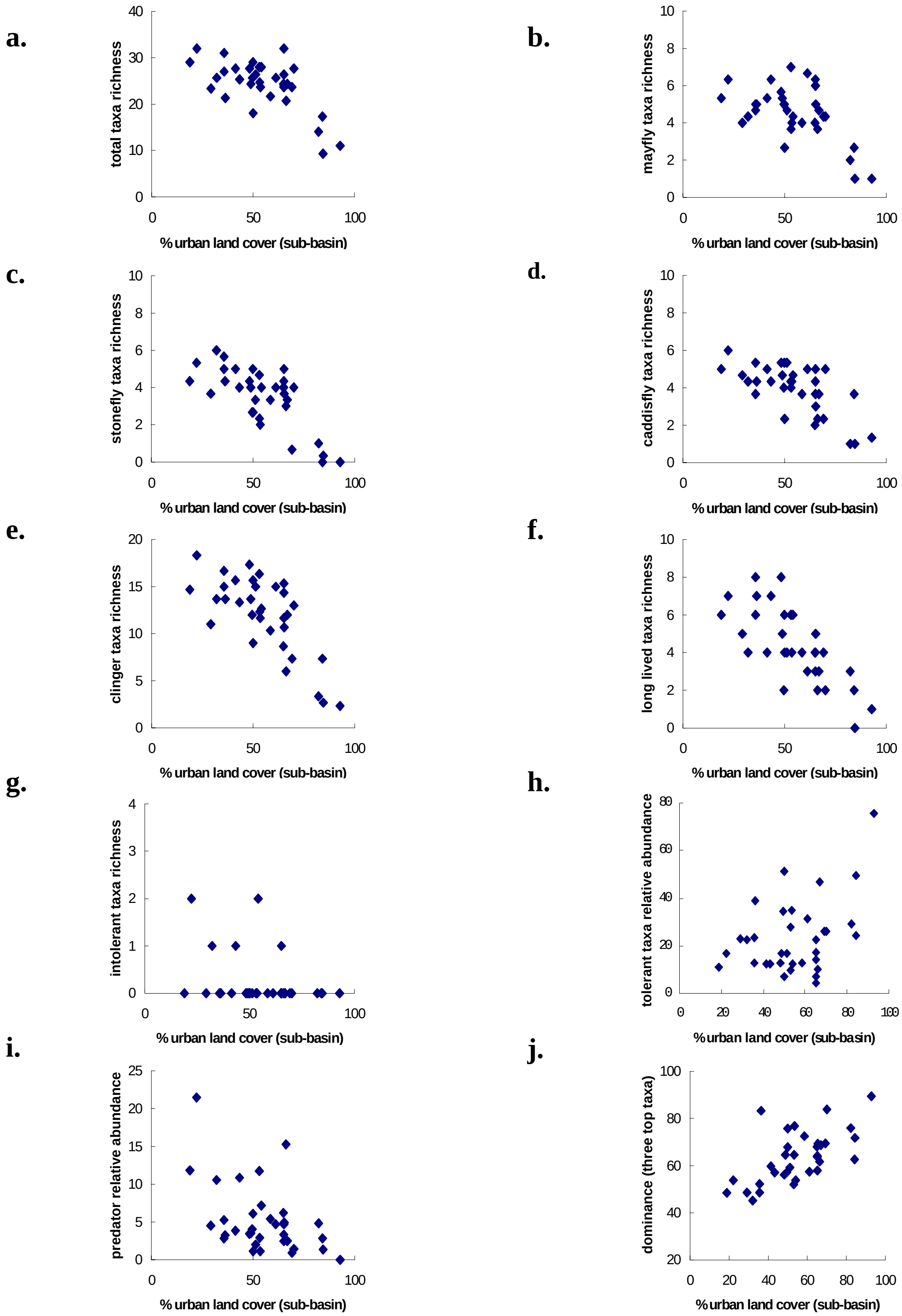

Figure 18. Relationship of B-IBI metrics to urban land cover (sub-basin). Note that for sites sampled in more than one year, only the most recent value is shown. Sites below restoration projects are also excluded from these graphs.

rural-urban gradient captured by the land cover classification. But urban streams are also very idiosyncratic; for instance, the extent of combined sewer overflow into streams may depend in part on how long ago a basin underwent urbanization. This study explores questions of spatial scale, but does not explicitly consider temporal scale effects. Some of the basins in this study (e.g., Thornton and Miller) have been urbanized for decades, while much of the outlying suburban development reflected in the 1998 satellite image is fairly recent (PSRC 1998). Along with the issue of old versus new infrastructure, some effects of recent land cover change may yet take years to be fully expressed in stream condition (Booth 1990, Harding et al. 1998). Examining long-term land cover change and more spatially complex models than considered here would enrich future study of the relationships between stream condition and urban development.

Conclusions and recommendations

The underlying goal of many urban stream management and restoration practices in the Pacific Northwest is biological. Instead of defining "critical thresholds" of basin development to generate formulas for stream protection, the biological condition of the streams that drain those basins should be examined directly. There is far too much complexity and uncertainty about ecological interactions to accurately predict biological condition in all cases with any land cover model—no matter how multi-factor or spatially sophisticated. While B-IBI was strongly related to the urban land cover measure generated from this study, this argues *not* for using one measure to predict the other, but for combining biological and landscape analysis to diagnose and address causes of stream degradation. Routine biological assessment is also critical for deciding how most effectively to spend limited restoration dollars. This study looked at a small sub-set of one type of restoration project: placement of large woody debris in small lowland streams. The results presented here can by no means be generalized across all types of restoration projects in the Pacific Northwest. But that is precisely the problem; at present, we don't know what's working and what is not so as to learn by example (Kershner 1997, Osborne et al. 1993). By ignoring the biology of those urban streams we seek to restore, we make the same mistake that has contributed to the current state of U.S. streams and rivers.

Monitoring guidelines.

1. Designate a portion of all restoration funds towards monitoring and assessment BEFORE the project even gets started.
2. Explicitly define restoration objectives: is the goal more wood, more pools, more salmon, healthy streams?
3. Design an appropriate monitoring program based on these project goals; i.e., don't limit monitoring efforts to counting pieces of wood or measuring plant survival rates if the underlying restoration goals are much broader than this.
4. Begin in-the-field monitoring efforts before restoration measures go into effect; baseline data are particularly critical when suitable control areas are not available.
5. Establish consistent sampling and analysis protocols for between year (and between site) comparisons; provide detailed descriptions of site locations.
6. Monitor at diverse spatial and temporal scales. Sample over time, and within and below projects to detect beneficial effects beyond project boundaries.
7. Incorporate volunteers into monitoring efforts as much as possible so as to foster a sense of "ownership" within the local community and to encourage long-term project maintenance.

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Appendix A. B-IBI study site locations (1997-1999)

Stream [1]	Site ID	Address (closest cross-streets)	Closest city	Site coordinates [2]	
				Lat.	**Long.**
Big Bear	BB971	Woodinville-Duvall Rd. & 210th Ave. NE	Woodinville	47.7579	122.0569
Big Bear	BB972	NE 164th St. & Mink Rd.	Redmond	47.7469	122.0596
Big Bear	BB973/981	NE 148th St. & Mink Rd.	Redmond	47.7364	122.0652
Big Bear	BB974	NE 148th St. & Mink Rd.	Redmond	47.7359	122.0657
Big Bear	BB975	NE 133rd St. & Beark Creek Rd.	Redmond	47.7183	122.0755
Big Soos	BS971	SE 290th St. & Kent - Black Diamond Rd.	Auburn	47.3407	122.1345
Forbes	FO98US	NE 106th Dr. & Forbes Creek Dr.	Kirkland	47.6967	122.1893
Forbes	FO98DS	108th Ave. NE & Forbes Creek Dr.	Kirkland	47.6961	122.1954
Jenkins	JE971	164th Pl SE & Covington-Sawyer Rd.	Covington	47.3462	122.1210
L.Jacobs	LJ99US	Sammamish Pkwy. SE & SE 43rd Wy.	Sammamish	NA	NA
L.Jacobs	LJ98US	Sammamish Pkwy. SE & SE 43rd Wy.	Sammamish	47.5649	122.0460
L.Jacobs	LJ99DS	Sammamish Pkwy. SE & SE 43rd Wy.	Sammamish	NA	NA
L.Jacobs	LJ98DS	Sammamish Pkwy. SE & SE 43rd Wy.	Sammamish	47.5654	122.0491
Little Bear	LB971	180th St. SE & 51st Ave. SE	Mill Creek	47.8336	122.1631
Little Bear	LB981	189th St. SE & 51st Ave. SE	Mill Creek	47.8264	122.1618
Little Bear	LB982	196th St. SE & 51st Ave. SE	Bothell	47.8197	122.1608
Little Bear	LB983 [3]	216th St. SE & 63rd Ave. SE	Bothell	47.8010	122.1497
Little Bear	LB972	228th St. SE & Hwy. 9	Woodinville	47.7909	122.1444
Little Bear	LB973/984	233rd Pl. SE & Hwy.9	Woodinville	47.7858	122.1449
Little Bear	LB974	233rd Pl. SE & 63rd Ave. SE	Woodinville	47.7819	122.1477
Little Bear	LB985	NE 195th St. & 136th Ave. NE	Woodinville	47.7728	122.1552
Little Bear	LB986	NE 177th Pl. & 134th Ave. NE	Woodinville	47.7587	122.1589
Little Bear	LB987	NE 178th St. & 130th Ave. NE	Woodinville	47.7560	122.1669
May	MA971	NE31st & Jones Ave.	Renton	47.5191	122.1937
Miller	MI971	168th Pl. SW & 8th Ave. SW	Normandy Park	47.4471	122.3475
North	NO981 [3]	183rd St. SE & John Bailey Rd.	Mill Creek	47.8344	122.2219
North	NO982	236th St. NE & Fitzgerald Rd.	Bothell	47.7804	122.1871
Rock	RO981 [3]	SE 262nd St. & Summit Landsburg Rd.	Maple Valley	47.3650	122.0136
Rock	RO971/982	SE 248th St. & Cedar River Pipeline Rd.	Maple Valley	47.3794	122.0197
Seidel	SE981	NE 133rd St. & 198th Ave. NE	Redmond	47.7185	122.0725
Soosette	SO99US	SE 304th St. & Hwy. 18	Auburn	NA	NA
Soosette	SO98DS	SE 304th St. & Hwy. 18	Auburn	NA	NA
Struve	ST981	NE 150th St. & 206th Ave. NE	Redmond	47.7336	122.0593
Swamp	SW981	164th St. SW & 28th Ave. W	Lynnwood	47.8509	122.2659
Swamp	SW982	181st Pl. SW & Butternut Rd.	Lynnwood	47.8321	122.2594
Swamp	SW983	Magnolia Rd. & Filbert Rd.	Lynnwood	47.8257	122.2553
Swamp	SW971	Larch Wy. SW & Locust Wy.	Brier	47.8109	122.2560
Swamp	SW972	Larch Wy. SW & Locust Wy.	Brier	47.8097	122.2566
Swamp	SW973/984	Larch Wy. SW & Locust Wy.	Brier	47.8090	122.2561
Swamp	SW985 [3]	Locust Wy. & Cypress Wy.	Brier	47.7995	122.2572
Swamp	SW986 [3]	Locust Wy. & Cypress Wy.	Brier	47.7993	122.2566
Swamp	SW987	Locust Wy. & Cypress Wy.	Brier	47.7991	122.2581
Swamp	SW988 [3]	Lockwood Rd. NE & Carter Rd.	Kenmore	47.7778	122.2496
Swamp	SW98US	NE 185th St. & 173rd Ave. NE	Kenmore	47.7661	122.2414
Swamp	SW99MS	NE 185th St. & 173rd Ave. NE	Kenmore	NA	NA
Swamp	SW98DS	NE 185th St. & 173rd Ave. NE	Kenmore	47.7635	122.2404
Swamp	SW989	NE 175th St. & 80th Ave. NE	Kenmore	47.7547	122.2343
Thornton	TH971	NE 107th St. & 15th Ave. NE	Seattle	47.7065	122.3136
Thornton	TH98US	NE 105th St. & 35th Ave. NE	Seattle	47.7066	122.2889
Thornton	TH98MS	NE 105th St. & 36th Ave. NE	Seattle	47.7056	122.2876
Thornton	TH98DS	NE 105th St. & 39th Ave. NE	Seattle	47.7053	122.2860

[1] basins are listed alphabetically and sites ordered from upstream to downstream
[2] units: degrees, datum: WGS84; [3] invertebrates collected but not processed at this site

Appendix B. Classification of invertebrates identified in study.

Phylum	Class	Order	Family	Genus	CL[1]	FFG[1]	T/I[2]	LL[2]
INSECTS		**Beetles**						
Arthropoda	Insecta	Coleoptera	Dystiscidiae	-		PR	T	LL
Arthropoda	Insecta	Coleoptera	Elmidae	Heterlimnius	CL	CG		LL
Arthropoda	Insecta	Coleoptera	Elmidae	Lara	CL	SH		LL
Arthropoda	Insecta	Coleoptera	Elmidae	Narpus	CL	CG		LL
Arthropoda	Insecta	Coleoptera	Elmidae	Optioservus	CL	CG	T	LL
Arthropoda	Insecta	Coleoptera	Elmidae	Zaitzevia	CL	CG	T	LL
Arthropoda	Insecta	Coleoptera	Haliplidae	Brychius	CL	MH	T	LL
Arthropoda	Insecta	Coleoptera	Hydrophilidae	Lacobius		PR		LL
		True Flies						
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogoninae		PR		
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyiinae		CG		
Arthropoda	Insecta	Diptera	Chironomidae	-		UN		
Arthropoda	Insecta	Diptera	Dixidae	Dixa		CG		
Arthropoda	Insecta	Diptera	Dixidae	Dixella		CG	T	
Arthropoda	Insecta	Diptera	Dixidae	Meringodixa		CG		
Arthropoda	Insecta	Diptera	Dolichopodidae	-		PR	T	
Arthropoda	Insecta	Diptera	Empididae	Chelifera		PR		
Arthropoda	Insecta	Diptera	Empididae	Clinocera	CL	PR		
Arthropoda	Insecta	Diptera	Empididae	Hemerodromia		PR	T	
Arthropoda	Insecta	Diptera	Empididae	Wiedemannia	CL	PR		
Arthropoda	Insecta	Diptera	Pelecorhynchidae	Glutops		PR	I	
Arthropoda	Insecta	Diptera	Phoridae	-		CG		
Arthropoda	Insecta	Diptera	Psychodidae	Maruina	CL	SC		
Arthropoda	Insecta	Diptera	Psychodidae	Pericoma		CG		
Arthropoda	Insecta	Diptera	Psychodidae	Psychoda		CG	T	
Arthropoda	Insecta	Diptera	Sciomyzidae	-		PR		
Arthropoda	Insecta	Diptera	Simuliidae	Simulium	CL	CF		
Arthropoda	Insecta	Diptera	Tipulidae	Antocha	CL	CG		
Arthropoda	Insecta	Diptera	Tipulidae	Dicronata		PR		
Arthropoda	Insecta	Diptera	Tipulidae	Hexatoma		PR		
Arthropoda	Insecta	Diptera	Tipulidae	Limnophila		PR		
Arthropoda	Insecta	Diptera	Tipulidae	Molophilus		UN		
Arthropoda	Insecta	Diptera	Tipulidae	Tipula		OM		
		Mayflies						
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella		CG	T	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis		CG	T	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Centroptilum		CG	T	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Diphetor		CG		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella	CL	CG		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	CL	CG	I	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	CL	CG		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	CL	CG		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Serratella	CL	CG		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Timpanoga	CL	CG		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygmula	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Heptagenia	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Ironodes	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Nixe	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Rhithrogena	CL	SC		
Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	Paraleptophlebia		CG		
Arthropoda	Insecta	Ephemeroptera	Tricorythidae	Tricorythodes		CG	T	
		Alderflies						
Arthropoda	Insecta	Megaloptera	Sialidae	Sialis		PR		
		Stoneflies						
Arthropoda	Insecta	Plecoptera	Capnidae	-		SH		

Appendix B. Continued.

Phylum	Class	Order	Family	Genus	CL[1]	FFG[1]	T/I[2]	LL[2]
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Kathroperla	CL	PR	I	
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Plumiperla	CL	PR		
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Sweltsa	CL	PR		
Arthropoda	Insecta	Plecoptera	Leuctridae	-		SH	I	
Arthropoda	Insecta	Plecoptera	Nemouridae	Malenka		SH		
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada		SH		
Arthropoda	Insecta	Plecoptera	Perlidae	Calineuria	CL	PR		LL
Arthropoda	Insecta	Plecoptera	Perlidae	Hesperoperla	CL	PR		LL
Arthropoda	Insecta	Plecoptera	Perlodidae	Isoperla	CL	PR		
Arthropoda	Insecta	Plecoptera	Perlodidae	Skwala	CL	PR		
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcella	CL	OM		LL
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcys	CL	OM		LL
		Caddisflies						
Arthropoda	Insecta	Trichoptera	Apataniinae	Apatania	CL	SC	I	
Arthropoda	Insecta	Trichoptera	Brachycentridae	Brachycentrus	CL	OM		LL
Arthropoda	Insecta	Trichoptera	Brachycentridae	Micrasema	CL	MH		
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Glossosoma	CL	SC		
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Protoptila	CL	SC		
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Cheumatopsyche	CL	CF	T	
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche	CL	CF		
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Parapsyche	CL	PR		LL
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila	CL	PH	T	
Arthropoda	Insecta	Trichoptera	Lepidostomatidae	Lepidostoma		SH		
Arthropoda	Insecta	Trichoptera	Limnephilidae	Ecclisomyia	CL	OM	I	
Arthropoda	Insecta	Trichoptera	Limnephilidae	Hydatophylax		SH		
Arthropoda	Insecta	Trichoptera	Limnephilidae	Onocosmoecus		OM		
Arthropoda	Insecta	Trichoptera	Limnephilidae	Psychoglypha		OM		
Arthropoda	Insecta	Trichoptera	Philopotamidae	Wormaldia	CL	CF		
Arthropoda	Insecta	Trichoptera	Polycentropodidae	Polycentropus	CL	PR		
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	CL	PR		
Arthropoda	Insecta	Trichoptera	Uenoidea	Neophylax	CL	SC		
NON-INSECTS:								
Arthropoda	Arachnida	Hydracarina	-	-		PA		
Arthropoda	Crustacea	Amphipoda	-	-		CG	T	
Arthropoda	Crustacea	Decapoda	-	-		OM		LL
Arthropoda	Crustacea	Copepoda	-	-		CG		
Arthropoda	Crustacea	Isopoda	-	-		CG	T	
Arthropoda	Crustacea	Ostracoda	-	-		CG		
Annelida	Hirudinea	-	-	-		PR	T	
Annelida	Oligochaeta	-	-	-		CG		
Coelenterata	Hydrazoa	-	-	-		PR	T	
Mollusca	Gastropoda	-	-	-		SC		
Mollusca	Pelecypoda	Veneroida	Sphaeriidae	-		CF		
Mollusca	Pelecypoda	Unionoida	Unionacea	-		CF		LL
Nematoda	-	-	-	-		OM		
Platyhelminthes	Turbellaria	-	-	-		CG		

[1] Clinger and functional feeding group classifications from Merrit and Cummins (1996)

[2] Tolerant/intolerant and long-lived classification from Wisseman (1999)

Appendix C. Data Dictionary

Layer	Geographic Extent	Source	Type	Resolution	Accuracy	Currency	% Complete
Streams	King, Pierce, Snohomish, and Kitsap counties with overlap into adjacent counties.	PRISM via King County via WA state Dept. of Natural Resources	line	1:48000	12 - 24m	9/23/94	NA
DEM	Sammamish-Cedar basin and Green River basin	PRISM via H. Greenberg via USGS	raster	10m	NA	NA	NA
Land cover	Portions of King, Pierce, and Snohomish counties	E. Botsford, K. Hill, and D. Booth via LandSat TM	raster	30m	NA	Aug-98	100%
Sample points	Portions of King and Snohomish counties	S. Morley via GPS unit	point	NA	5 - 10m	3/1/99	89%
	All map units in meters, projection UTM, Zone 10, Datum NAD27						

Appendix D. B-IBI raw metric and total scores for study sites.

Site ID	Total Taxa	Ephem.	Plecop.	Trichop.	Dom	L. Lived	Intol.	% Tol.	Clinger	% Pred.	B-IBI
1997											
BB971	23.3	4.0	3.7	4.7	48.6	5	0	23.1	11.0	4.5	**32**
BB972	25.7	4.3	6.0	4.3	45.1	4	1	22.4	13.7	10.6	**36**
BB973	22.3	4.0	5.0	4.0	56.2	5	0	11.4	12.7	12.5	**32**
BB974	27.0	5.0	5.7	3.7	52.2	6	0	23.6	15.0	5.3	**34**
BB975	27.7	5.3	5.0	5.0	59.7	4	0	12.2	15.7	3.9	**28**
BS971	25.7	6.7	4.0	5.0	57.5	3	0	31.5	15.0	4.7	**26**
JE971	25.3	6.3	4.0	4.3	57.0	7	1	12.5	13.3	10.9	**32**
LB971	28.0	4.3	4.0	4.7	53.8	6	2	12.3	12.7	7.2	**36**
LB972	24.3	5.3	4.0	4.7	64.6	5	0	16.7	13.7	3.5	**28**
LB973	19.7	4.7	2.3	4.0	55.3	2	0	31.5	10.3	2.3	**22**
LB974	18.0	2.7	2.7	2.3	75.8	4	0	51.4	9.0	1.1	**16**
MA971	21.3	5.0	4.3	4.3	83.3	7	0	38.8	13.7	3.3	**24**
MI971	9.3	1.0	0.3	1.0	71.8	0	0	49.4	2.7	1.3	**12**
RO971	33.3	7.0	5.7	6.7	53.1	9	3	9.3	19.7	11.1	**48**
SW971	26.3	6.3	4.3	4.3	58.0	4	0	22.6	14.3	3.3	**28**
SW972	23.7	6.0	4.0	3.7	64.1	3	0	14.1	11.7	2.5	**26**
SW973	30.7	6.3	5.0	4.3	77.0	6	0	24.1	15.3	2.3	**28**
TH971	13.0	1.5	0.5	0.0	53.7	2	0	61.5	1.5	0.6	**14**
1998											
BB981a	31.00	4.67	5.00	5.33	48.65	8	0	12.62	16.67	2.83	**38**
FO98ds	13.00	1.00	0.00	1.33	74.81	3	0	14.52	3.00	0.81	**16**
FO98us	14.00	2.00	1.00	1.00	75.93	3	0	29.12	3.33	4.82	**16**
LB981	28.00	7.00	4.67	4.33	51.96	6	0	9.90	16.33	11.75	**40**
LB982	29.00	5.00	5.00	5.33	67.89	6	0	6.89	15.67	6.11	**34**
LB984	25.67	5.00	2.67	4.00	57.15	2	0	34.69	12.00	4.06	**22**
LB985	26.33	4.67	3.33	5.33	59.21	4	0	16.71	15.00	2.00	**30**
LB986	24.67	3.67	2.33	4.00	64.64	6	0	27.91	12.33	2.92	**24**
LB987	23.67	4.00	2.00	4.33	76.78	4	0	34.98	11.67	1.14	**22**
LJ98ds	21.67	4.00	1.33	3.67	62.47	2	0	32.69	7.33	4.69	**22**
LJ98us	21.67	4.00	3.33	3.67	72.47	4	0	12.90	10.33	5.41	**30**
NO982	24.33	4.67	3.33	3.67	68.70	3	0	46.92	12.00	2.50	**22**

Appendix D. Continued.

Site ID	Total Taxa	Ephem.	Plecop.	Trichop.	Dom	L. Lived	Intol.	% Tol.	Clinger	% Pred.	B-IBI
RO982	32.00	6.33	5.33	6.00	53.74	7	2	16.69	18.33	21.47	**44**
SE981	29.00	5.33	4.33	5.00	48.49	6	0	10.93	14.67	11.82	**36**
SO98ds	34.00	7.67	5.67	5.67	49.54	5	3	14.86	18.00	11.07	**46**
ST981	27.67	5.67	4.33	5.33	56.18	8	0	12.81	17.33	3.44	**34**
SW981	20.67	3.67	3.00	2.33	61.75	2	0	10.10	6.00	15.28	**26**
SW982	24.33	4.00	4.00	2.00	67.97	4	0	4.35	8.67	6.22	**28**
SW983	24.00	5.00	3.67	3.00	69.34	5	0	7.12	10.67	4.95	**30**
SW984	32.00	6.00	5.00	5.00	63.64	5	1	17.26	15.33	4.72	**32**
SW987	17.33	2.67	0.00	3.67	62.63	2	0	24.12	7.33	2.83	**20**
SW989	23.67	4.33	0.67	2.33	69.47	4	0	25.97	7.33	0.92	**22**
SW98ds	30.67	5.00	4.00	5.00	73.13	3	0	38.20	13.33	1.99	**26**
SW98us	27.67	4.33	4.00	5.00	83.79	2	0	25.88	13.00	1.43	**24**
TH98ds	15.67	1.00	0.00	1.33	83.51	1	0	74.13	3.00	0.36	**12**
TH98ms	12.33	1.00	0.00	2.33	88.85	1	0	67.63	3.33	0.09	**10**
TH98us	11.00	1.00	0.00	1.33	89.43	1	0	75.67	2.33	0.00	**10**
1999											
LJ99ds	25.30	5.00	3.00	4.00	60.10	4	0	17.35	11.00	9.89	**32**
LJ99us	22.00	4.00	4.00	3.67	71.00	6	2	25.63	12.00	5.39	**32**
SO99us	32.30	6.67	4.67	5.33	53.10	5	1	11.98	15.00	7.91	**36**
SW99ms	29.00	5.67	3.00	4.33	58.36	3	0	33.62	11.67	3.17	**26**